

Natural Range of Variation (NRV) for yellow pine and mixed conifer forests in the bioregional assessment area, including the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests.

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INTRODUCTION

Natural Range of Variation (NRV) assessments (essentially equivalent to Historical Range of Variation [HRV] assessments) provide baseline information on ecosystem conditions (composition, structure, and function) that can be compared to current conditions to develop an idea of trend over time and an idea of the level of departure of altered ecosystems from their “natural” state (Morgan et al. 1984, Manley et al. 1995, Landres et al. 1999, Wiens et al. 2012; see Methods). These trend assessments form part of the basis for the assessment of ecological integrity that is required in the 2012 Forest Service Planning Rule. NRV assessments were carried out for 11 terrestrial ecosystems by the Pacific Southwest Region Ecology Program between October, 2012 and May, 2013, using historical information (primarily from the pre-Euroamerican period, 16th century to the mid-19th century) as well as information from modern-day reference ecosystems and other sources (see Methods and Appendix).

In the Pacific Southwest Region (“Region 5”), Forest planning under the 2012 Planning Rule is currently intended to occur in geographically contiguous groups of National Forests, at intervals of two to three years. Each Forest will carry out its own set of assessments, but Region 5 also decided to carry out a bioregional assessment covering the same area as the 1996 Sierra Nevada Ecosystem Project (SNEP), so as (1) to update trends and conditions described in the SNEP report and the Sierra Nevada Forest Plan Amendment (USDA 2004), and (2) to better provide consistency among the National Forest plans with respect to conditions and trends at spatial scales greater than the size of a single National Forest (400,000 to 800,000 hectares), and temporal scales greater than the typical lifespan of a Forest Plan (15-20 years). In this report, we provide (1) an estimate of the NRV for key elements of yellow pine and mixed conifer forest ecosystems, including quantitative ranges for specific variables whenever possible; and (2) a comparison of our NRV analysis with current conditions in the bioregional assessment area. Our NRV assessments also go further in time, and summarize the current science concerning projected future conditions for key ecosystem elements.

Physical setting and geographic distribution

Yellow pine and mixed conifer forests are found throughout the assessment area (Figure 1). In this chapter, we jointly consider the two forest types and refer to them collectively as “YPMC” forests (reasoning behind this combination is given in the Ecological Setting section, below). YPMC forests are the most widely distributed vegetation type in the assessment area. Based on the California Wildlife Habitat Relations (CWHR) vegetation types, forest types corresponding to YPMC forest cover about 3 million hectares (7.5 million acres) of the assessment area (Figure 1).

YPMC forests are found on a variety of soils and bedrocks and there is little evidence of strong soil chemistry-driven differences in plant species composition or forest structure in the assessment area, except in the case of ultramafic (“serpentine”) soils, which are extremely nutrient deficient but relatively rare in the YPMC forest belt, aside from parts of the northern Sierra Nevada (Alexander et al. 2007). On the other hand, variations in soil depth and texture, which are strongly related to topography (slope, aspect, elevation, slope shape, etc.) in the assessment area’s Mediterranean-type climate, are major drivers of variation in forest species composition,

density, cover, and other related variables. A major role of soils in assessment area YPMC forests is their contribution to water availability (O'Geen et al. 2007).

Most of California, and the assessment area, falls with the North American Mediterranean-climate zone. Five geographic areas on earth share the unique Mediterranean climatic characteristics, which are typically expressed on the western boundaries of continents between about 30° and 45° latitude. Under the Köppen (1931) classification, the Mediterranean climate ("Cs" in the Köppen classification) is described as a temperate rainy climate with hot, dry summers. As Bailey (2009) notes, the combination of wet winters with dry summers is unique along world climate types, and leads to a distinctive vegetation dominated by evergreen trees and shrubs. The major environmental stress is the severe summer drought, which can last 3-6 months, and high evaporative demand during much of the growing season. In the Mediterranean-type climate, the demand for water and its supply are exactly out of phase (Major 1988). Within the Cs climate zone, there is a gradient in intensity of the dry summer season from areas with relatively mild summer temperatures (northern assessment area and higher elevations) to areas with hot summer temperatures (lower elevations, especially in the southern and eastern assessment areas) (Specht 1982).

Figure 2 shows Walther type climate diagrams for six stations in or near YPMC forests in the assessment area; data are from WRCC (2013). The diagrams are organized to approximately match their locations in the map in Figure 1. As per Walter and Lieth (1967), the temperature and precipitation axes are scaled such that 20 mm precipitation = 10°C temperature. The dry season is found (approximately) where the precipitation line undercuts the temperature line; the wet season occurs where precipitation supercedes temperature. We say approximately because moisture stored in the soil is available for plant use after the end of the wet season and can reduce true dry season length by a month or more depending on annual precipitation (see Major 1988).

The climate diagrams underline the major climatic gradients in the assessment area (Figure 2). Dry season length is shortest along the west slope, especially in the north (3-4 months), and longest in the extreme south and on the east slope (5-6 months). Precipitation is higher in the northwest than the rest of the assessment area. Stations east of the hydrologic divide (which is either the Sierra Nevada or Cascade Range crest in much of the assessment area) are subject to a rain shadow effect, as is the Modoc Plateau, represented by Alturas. Stations with moderate Great Basin influence (e.g. Alturas and Lee Vining) receive some monsoon-derived precipitation in the summer months. Temperatures are warmest in the south, and coolest in the north and at higher elevations (Figure 2).

Topography drives major differences in ecosystem distribution across the assessment area. Elevation increases from north to south, and local relief is generally much higher in the central and southern Sierra Nevada than any of the other subregions. The Sierra Nevada is also narrower in the south, and the combination of less land area, higher mountains, and steeper slopes means that the YPMC forest belt is much more compressed in the southern subregion (Figure 1). Steeper terrain, a drier climate and the predominance of granitic bedrock in the central and south subregions also lead to much greater prevalence of bare rock substrates and cliffs. Overall, the natural lay of the land means that physical barriers to ecological processes like animal

migration, plant seed dispersal, and fire spread are greater in the southern and central subregions than elsewhere, but land ownership and land use patterns follow the opposite pattern, with much more fragmentation of habitat in the northern subregion.

Because of dry growing season conditions in the assessment area, water availability is a major driver of ecosystem distribution and condition (Major 1988, Loik et al. 2004, Barbour et al. 2007). Topography exerts strong influence on water availability, both directly, through its influence on solar insolation and evaporation, and indirectly, through its influence on soil depth and texture. Forest conditions in the YPMC forest belt vary substantially along topographic gradients (Sugihara et al. 2006, Barbour et al. 2007, North et al. 2012b). It can be generalized that, all other environmental factors being equal, YPMC forest cover and density tend to be higher on north-facing (“cool”) aspects (where water availability is higher), and more open and less dense on south- and west-facing (“warm”) aspects. Because higher elevations lose less water to heat evaporation, they also tend to support denser stands of forest than lower elevations. The upper portions of mountain or canyon slopes lose substantial water to gravity-driven flow and tend to support more open forest conditions, whereas lower slopes are net importers of water from higher slopes, and they support denser forest conditions. The most open forest conditions tend to be on south- or west-facing, upper, convex canyon slopes at lower elevations; the densest forest conditions tend to be on north-facing, concave, lower slopes. Overlain on these topographic effects are precipitation and temperature gradients across the broader assessment area. In general, forests are more open (and more composed of drought-tolerant species) in the southern and eastern portions of the assessment area than in the western and, especially, the northern portions.

Ecological setting

Yellow pine forests in the assessment area are those that are dominated by one or both of the “yellow pine” species present in California: ponderosa pine (*Pinus ponderosa*), and Jeffrey pine (*P. jeffreyi*); Washoe pine is now considered a subspecies of ponderosa pine (*Pinus ponderosa* ssp. *washoensis*; Baldwin et al. 2012). Ponderosa and Jeffrey pine are closely related (they are both in the subgenus *Pinus*, section *Pinus*, subsection *Ponderosae*) and they occasionally hybridize. Ponderosa pine, the most widely distributed pine species in North America, is found throughout the mountainous regions of California, whereas Jeffrey pine is primarily a California tree, with some occurrences in westernmost Nevada, southwestern Oregon and northern Baja California. Of the two species, Jeffrey pine is more stress tolerant, and replaces ponderosa pine at higher elevations, on poorer soils, and in colder and/or drier climates (Haller 1959, Barbour and Minnich 2000). Ponderosa pine dominated forests can occur from about 300 m elevation to about 1800 m in the northern subregion of the assessment area, and from about 1200 m to 2100 m in the southern subregion (Fites-Kaufman et al. 2007); Jeffrey pine dominated forests occur mostly between 1500 and 2400 m in the northern subregion and from 1700-2800 m in the southern subregion (with the highest elevations usually being on the east side of the Sierra Nevada; Fites-Kaufman et al. 2007, Barbour and Minnich 2000). Both yellow pine species can also occur in other forest types. A large area of the Lassen, Plumas, and Tahoe National Forests east of the Sierra Nevada crest supports a mixed yellow pine forest with codominance by ponderosa and Jeffrey pine; this forest type is often called “eastside pine”.

Many conifer species with overlapping geographic distributions are found in the assessment area, and difficulties in differentiating obvious forest types have led to the general recognition of a “mixed conifer” belt in the lower montane zone, usually intermixed with yellow pine dominated stands at its lower edge and up to 2000-2200 m elevation at its highest, depending on latitude. Major tree species include the yellow pine species, sugar pine (*Pinus lambertiana*), white fir (*Abies concolor*), incense cedar (*Calocedrus decurrens*), Douglas-fir (*Pseudotsuga menziesii*), and black oak (*Quercus kelloggii*). Red fir (*Abies magnifica*), lodgepole pine (*Pinus contorta*), and western white pine (*Pinus monticola*) are upper montane species that also make appearances in higher mixed conifer stands; a variety of hardwood species also occur in lower elevation stands, including canyon live oak (*Quercus chrysolepis*), interior live oak (*Q. wislizenii*), and tan oak (*Lithocarpus densiflorus*). As Barbour and Minnich (2000) note, the yellow pine species (especially ponderosa pine) are “the biological thread that holds the (mixed conifer) forest together”, but the very fine-grained pattern of local dominance makes it difficult to clearly discriminate the various “phases” for classification or mapping purposes.

Although Sawyer et al. (2008) describe nearly 20 different forest alliances that make up the montane mixed conifer forest, we stick with common practice and treat the mixed conifer forest as a single, if geographically mutable, entity. In some cases in this assessment we refer to “dry mixed conifer” or “moist mixed conifer” forests. These terms are used only in a general sense, and differentiate mixed conifer based on the yellow pine and fir components (yellow pine dominance in dry mixed conifer, greater fir presence in moist mixed conifer) and the annual precipitation (mostly <1000 mm in dry mixed conifer, mostly >1000 mm in moist mixed conifer); moist mixed conifer stands are also more common at higher elevations. We also follow Barbour and Minnich (2000) in combining yellow pine forest and mixed conifer in this chapter. This is because yellow pine forests have become rare on the west side of the Sierra Nevada, due to logging of the ponderosa pine and/or fire suppression-driven succession to mixed stands dominated by more shade tolerant and fire intolerant species like white fir and incense cedar. Indeed, contemporary “dry mixed conifer” stands may to a great extent represent formerly yellow pine dominated stands that have experienced infill subsequent to logging and fire suppression. Yellow pine dominated forests remain common on the east side of the Sierra Nevada, however.

We do not explicitly treat giant sequoia (*Sequoiadendron giganteum*) in this NRV assessment. Giant sequoia occurs in scattered groves in the southern assessment area, with a few occurrences in the central assessment area. Giant sequoia is a locally dominant member of moist mixed conifer stands with particular topographic and soils conditions (Sugihara et al. 2006, Barbour et al. 2007). Our treatment of moist mixed conifer forest, which is otherwise dominated by species like white fir and sugar pine, is generally applicable to giant sequoia stands, but we direct the reader to the forest-level assessments for more detail. Stephenson et al. (1999) is an excellent treatment of ecological, management, and restoration issues in giant sequoia-dominated mixed conifer forest.

An understanding of past, current, and possible future conditions in YPMC forests in the assessment area requires consideration of the ecological differences between the major tree species. Forestry experience and scientific investigation have provided us with a wealth of information vis-à-vis the relative ecological tolerances of major tree species in the YPMC forests of the assessment area. Table 1 offers some summary information about geographical and elevational

distributions of the seven major tree species in assessment area YPMC forests. Four of the seven species are broadly restricted to the North American Mediterranean-zone (Jeffrey pine, sugar pine, incense cedar, black oak), and three are more widely distributed (ponderosa pine, Douglas-fir, white fir), with the two former species among the most widely distributed tree taxa in North America. Species restricted primarily to the YPMC forest zone include ponderosa pine, sugar pine, white fir, incense cedar, and black oak. Douglas-fir is also found at lower elevations, usually in moist, mixed evergreen forests with a number of hardwood/broadleaf associates. Jeffrey pine ranges into higher elevation forests and is a common member of red fir forests and even some warm sites in the subalpine zone (Table 1).

We have compiled and condensed important information related to comparative ecological traits and tolerances for the important tree species in assessment area YPMC forests. Table 1 provides information on seed weights, which are inversely related to dispersal distances (and therefore scale with the relative rapidity by which these species can recolonize or migrate to distant habitat), and leaf flammability, which is a trait strongly associated with the importance of fire to species ecology (Fonda et al. 1998, Keeley and Zedler 1998, Keeley et al. 2012). Table 2 ranks tree species in assessment area YPMC forests by their relative tolerances to shade, frost, temperature, drought, and fire. Figure 3 compares growth rates of young shade tolerant and intolerant tree species, and Figure 4 compares bark thickness of young trees of the six major conifer species from Table 1.

McCune (1988) defined groups of North American pines that could be reliably differentiated based on their ecological characteristics. Ponderosa, Jeffrey and sugar pine all formed part of his “fire resisters” species group, which was characterized by traits and tolerances that promoted the ability to survive fire. This group was generally marked by long needles, thick twigs (which protect growing tissues), thick bark, thick cone scales (which may protect seed from heat), relatively slow growth, and an extended adolescent period (the mean time to seed-producing age was 16.3 years). McCune (1988) noted that this group of pines was tolerant of frequent surface fire “and reacts to fire more as a stress than as a disturbance.”

Bark thickness of young YPMC tree species as a function of tree diameter is shown in Figure 4. Among YPMC tree species, the two yellow pine species have the thickest bark at young ages, but are passed by incense cedar at about 10 cm dbh and other species between 25 and 50 cm dbh (Dolph 1984; also see van Mantgem and Schwartz 2003); it should be noted that, its thick bark notwithstanding, mature incense cedar is somewhat more susceptible to cambial injury than its bark thickness might indicate, due to the bark’s dry, stringy, and highly furrowed nature (Lachmund 1923). Yellow pines are thus well protected from fire during the sapling stage, which is critical in an ecosystem characterized by fires every 5-20 years on average (Van de Water and Safford 2011). Other fire-related traits of ponderosa, Jeffrey, and sugar pine include self-pruning of lower branches (Keeley and Zedler 1998) and highly flammable needles and cones, which promote fires that kill competitors that are less fire tolerant (Fonda et al. 1998, Fonda and Varner 2004). Black oak, which is a common member of drier YPMC forests in the assessment area, also produces extremely flammable litter (it is the most flammable western US oak; Engber and Varner 2012). Douglas-fir is an interesting case. Young Douglas-fir are quite susceptible to fire mortality, but mature individuals can be extremely fire resistant (Agee 1993, Skinner et al. 2006). The species develops very thick bark when mature, and the bark is less prone to slough-

ing and collecting at the base of the tree than pine bark. Douglas-fir needles are also short and relatively inflammable (Fonda et al. 1998; see below). It has been hypothesized that Douglas-fir's presence in frequent-fire forests was favored by a highly variable fire frequency over time, one that periodically included fire-free intervals sufficiently long to allow recruitment of Douglas-fir seedlings into age classes that are more fire resistant (Agee 1993).

Fonda et al. (1998) tested flammability and burning characteristics of the needles of 13 common conifer species from the western US. Based on six burning characteristics, ponderosa and Jeffrey pine were ranked numbers one and two, sugar pine was number seven (giant sequoia was number six). Douglas-fir and the two tested fir (*Abies*) species were three of the four lowest ranked species (Fonda et al. 1998). Fonda et al. (1998) noted that the highest ranked species were characterized by fire regimes supported by nonwoody fuels (e.g., needle litter, herbaceous fuels), while the lower ranked species typically required woody fuel accumulation over decades in order to support fires. In another flammability study, de Magalhaes and Schwilk (2012) found that black oak, ponderosa and Jeffrey pine had the highest flammability across most parameters tested. White fir and red fir were less flammable, but they were both more flammable than the fir species tested by Fonda et al. (1998), which is not surprising given their presence in the highly fire prone Sierra Nevada. de Magalhaes and Schwilk (2012) also found that flammability of species mixtures of needles was mostly driven by the most flammable species in the mixture, which were Jeffrey, ponderosa, and sugar pine, and black oak. See Table 2 for a fire tolerance ranking of assessment area species.

Major tree species in YPMC forests differ notably with respect to the size of their seeds (Table 1), the distances those seeds are dispersed, and the numbers of seeds they produce. In general, heavier seeded species disperse shorter distances, although animal vectors play an important role in the dispersal of some species. Fowells and Schubert (1956) measured seed rain in a YPMC forest during a nine year period and found that, on average, white fir produced about 2.5 times as much seed as either ponderosa or sugar pine, and incense cedar produced about four times as much seed. Fowells and Schubert (1956) found very high interannual variability, and in some years the pines outseeded the two shade tolerant species. Van Mantgem et al. (2006), working in Yosemite and Sequoia-Kings Canyon National Parks, concluded from a five year study that, standardized by basal area, fir species produced between seven and eight times as much seed as pines (yellow and sugar), and incense cedar produced about three times as much seed. Zald et al. (2008) found seed rain of shade-tolerant white fir and incense cedar was 5–26 times greater than Jeffrey and sugar pine in southern Sierra Nevada mixed conifer. Stark (1965), in a study of natural tree regeneration after logging, reported that over a 13 year period, over 80% of surviving seedlings were incense cedar or white fir, 10% sugar pine, and 4.3% yellow pine. Fowells and Schubert (1956) found that white fir and ponderosa pine had similar dispersal distances, which were 50-75% further than sugar pine. Vander Wall (2003) found that rodents and birds were important dispersers of pine seeds in YPMC forests. Rodents moved seeds of ponderosa, Jeffrey, and sugar pine about 25 m on average from the parent plants, which approximately doubled the dispersal distances for the latter two species.

Growth rates of YPMC forest trees vary by species and environmental conditions. Yeh and Wensel (2000) found that diameter growth of assessment area YPMC species typically occurs between late March/early April and mid September, but growth ends earlier at lower elevations (due to

summer drought) and starts later at high elevations. Yeh and Wensel (2000) found that the pines were less disadvantaged by water stress than other tree species. Overall the most important factors to growth were tied to water availability, and included the previous winter's precipitation (related to water storage) and the current summer temperature (related to water loss).

Light availability is an important environmental driver of growth for plants, and YPMC tree species differ notably in the effects that canopy shading has on their growth rates (Table 2). Keyser (2010) provides equations for growth of major tree species in the assessment area. Small tree (dbh between 3.8 and 7.6 cm [1.5 and 3"]) growth is modeled as a function of site productivity (site index, or the mean potential height in feet of a free-grown tree at 100 years), crown ratio (ratio of crown length to tree height), and competition/shading by larger trees (basal area of trees larger than the subject tree) (Keyser 2010). Figure 3 shows how both the yellow pine species and sugar pine grow more rapidly than white and red fir in open stands, but are out-grown by the fir species once basal area of larger trees exceeds about 30 m²/ha (130 ft²/ac). Bigelow et al. (2012) examined the crossover-point irradiance (CPI), the light at which the height growth rank of pairs of species changes, and found that an understory light environment with 41% or more of full sunlight favored ponderosa pine regeneration over white fir. Moghaddas et al. (2008) showed that black oak and ponderosa pine seedlings seedling survival was strongly connected to high light environments. Oliver and Dolph (1992) found that ponderosa pine grew more slowly than the other YPMC tree species at even moderate shade levels, while sugar pine showed quite strong growth at 50-60% of full sunlight. They noted that sugar pine appeared to be adapted to exploit small forest gaps, and showed some characteristics of the more shade tolerant species (e.g., fir).

YPMC tree species also differ with respect to their tolerances to frost, heat and cold, and drought. In general, shade tolerant species from low and middle elevation forests (e.g., Douglas-fir and white fir) are more susceptible to frost damage than the pine species and red fir (Table 2). Not surprisingly, the most cold tolerant species are those from higher elevations and the most heat tolerant are those from lower elevations and warmer microsites (Table 2). Drought tolerance is a very important trait, as precipitation in the assessment area is highly variable between years, and models of future climate in the assessment area project increased occurrence of drought in the future (Liu et al. 2010, Dettinger 2011). The true fir species are the least drought tolerant members of YPMC forests in the assessment area, while sugar pine, Douglas-fir, and incense cedar are intermediate. The yellow pines and black oak are the most drought tolerant species in YPMC forests (Table 2).

In summary, the major YPMC tree species are differentially adapted to the physical and biotic environment in the assessment area, and the different tolerances of these tree species play a major role in determining forest composition, structure and function. Considering the overwhelming importance of fire and water availability to YPMC forest ecology and management in the assessment area, perhaps the most important distinction to be made is between those species that are highly tolerant of fire and drought but intolerant of shade (black oak and the yellow pines), versus those that are less tolerant of fire and drought but grow relatively well in low light conditions (white fir, incense cedar, Douglas-fir; recall however the high fire tolerance of mature individuals of the latter species). These ecological differences are at the root of general topographic tendencies in forest composition across the assessment area. Locations with higher

water availability (north aspects, lower slopes, concave topography, higher elevations) tend to support higher densities of the shade-tolerant, fire-intolerant species, whereas sites with lower water availability (south aspects, upper slopes and ridgetops, convex topography, lower elevations) are more likely to be dominated by shade-intolerant, fire-tolerant species.

These ecological differences among species translate into successional differences as well. In regions that can support forest cover, trees that specialize on high light environments will be generally restricted to early successional stages, and locations where ecological factors slow successional processes. The very high frequency of fire before Euroamerican settlement (see below) played such a role, and essentially kept YPMC forest in the assessment area in a state of arrested development, where early successional species such as the yellow pines and black oak were able to maintain canopy dominance, even in places of relatively high water availability. In the general absence of fire disturbance over the last century, successional processes have become “unblocked”, and more competitive, later successional species like the firs and incense cedar have come to dominate most of the YPMC belt in the assessment area. Early successional species continue to dominate where fire regimes have not been much perturbed, and where local soil productivity acts to slow succession. We will make reference to these differences in fire, shade, and drought tolerance and successional relationships throughout this chapter.

Temporal variability in the ecological setting

The mutable nature of the climate has never been more apparent than today, with human inputs to the atmosphere rapidly increasing greenhouse levels and global temperatures. Although the concept of “climate” is one that suggests long-term stability, climates are constantly changing, and climatic variability throughout the Holocene Epoch has had major effects on YPMC forests in the assessment area. It is also important to note that temporal changes in climate have not been uniform across the assessment area landscape, and the timing of changes in temperature and precipitation, as well as biotic responses, differ from one area to another. Nonetheless, some useful generalizations can be made. The Introduction to the NRV assessments contains more detail about Holocene climatic variability, but a brief overview is warranted here.

The Holocene Epoch is now considered to have begun about 12,000 ybp (years before present). The entire epoch falls within a broadly defined “interglacial” period. The earth’s climate has been in “glacial” periods for about 90% of the last 850,000 years, and interglacial periods as warm as the Holocene are relatively rare (Tausch et al. 1993). Overall, mean annual temperatures have fluctuated by 3 to 6 degrees C through the Holocene, and precipitation has also risen and fallen. Researchers generally divide the Holocene into three periods, which are defined by broad (and somewhat ill-defined) changes in temperature and precipitation. The Early Holocene stretches from the beginning of the epoch to 8000 or 7000 ybp, and was characterized by post-glacial warming, but generally cool and moist conditions compared to today. Many closed basins east of the Sierra Nevada crest supported large rain- and snowmelt-fed lakes at the beginning of the Holocene, but most of these dried or decreased greatly in size over the ensuing couple of millennia (Minnich 2007). At the beginning of the Holocene, elevations that currently support YPMC forests were largely vegetated by high elevation sagebrush and grass species, with minor presence of pines and juniper (Woolfenden 1996). By 9000-10000 ybp however, conifer forests had established themselves in most of these areas (Minnich 2007).

The Middle Holocene, about 8000 to 4000 ybp, is also often referred to as the “Xerothermic”, “Hypsithermal”, or “Altithermal” Period. Climates became much warmer and drier, with the driest and warmest conditions occurring around 6000 ybp. Glaciers completely disappeared from the Sierra Nevada (Clark and Gillespie 1997), and lake levels dropped precipitously. For example, Lake Tahoe was apparently permanently below the Truckee River outlet elevation during the period between about 6300 and 4800 ybp (Lindström 1990). Fire frequency also increased during the Middle Holocene, at least in places that were able to maintain plant cover (Woolfenden 1996, Beaty and Taylor 2009). Paleoecological data suggest that forests of fir and pine were replaced by oak, sagebrush, and juniper in many areas, and forest structure was likely very open, with abundant understory shrubs. Conifers invaded former moist areas of meadow, and desert plant and animal taxa migrated upslope (Anderson 1990, Minnich 2007).

The Late Holocene (4000 ybp to present) has been generally characterized by cooling, with some warmer periods. Precipitation increased, and small glaciers began to form again in the Sierra Nevada. Millar and Woolfenden (1999) suggest that the basic spatial and compositional outlines of modern Sierra Nevada ecosystems developed by the beginning of the Late Holocene. As temperatures cooled, available moisture rose, and fir and incense cedar abundance increased relative to pine and oak; giant sequoia began to colonize their current groves (Millar and Woolfenden 1999). White fir was a mostly minor component of assessment area YPMC forests before the general cooling and increase in precipitation seen over the last 4,000 years. Oaks have declined in importance since the end of the Middle Holocene. Earlier in the Holocene, sagebrush was a major component of low elevation west side landscapes, but now it is primarily restricted to higher elevations on the east side of the assessment area. Fire has been present as an important ecosystem disturbance ever since deglaciation at the beginning of the epoch, but the “presettlement period” characterized by very frequent fire and large areas of “fire-adapted” vegetation began during the Late Holocene (Millar and Woolfenden 1999).

The last 1000 years of the Holocene have been marked by short term changes in temperature and precipitation that have had major impacts on assessment area ecosystems (Woolfenden 1996, Millar and Woolfenden 1999, Minnich 2007). Between about 900-1100 AD and 1200-1350 AD, two long drought periods (“Medieval Droughts”, or collectively, the “Medieval Warm Period”) led to very low levels in lake and streams and increased fire frequencies. This was followed by a shift to cooler temperatures known as the “Little Ice Age”, caused apparently by a series of massive volcanic eruptions that caused atmospheric reflection of solar radiation (Miller et al. 2012). The Little Ice Age lasted from about 1400-1880, and the period between 1650 and 1850 was the coolest since the Early Holocene (Stine 1996). Glaciers expanded in the Sierra Nevada, treeline dropped, and fire frequencies moderated. Minnich (2007) notes however that there is little evidence for major vegetation change during the Little Ice Age. It is important to underline that the period of the Little Ice Age is also the period most commonly used as an historical reference period for restoration planning in the western US.

Most recently, human emissions to the atmosphere have resulted in renewed warming, even though earth orbital cycles should be resulting in a cooling trend (Ruddiman 2005). Current temperature trends include increased temperatures (the western US is heating faster than any other part of the country) especially at nighttime, a decrease in the number of days with below freezing temperatures, and an increase in the number of extreme heat days. The 20th century

was one of the wettest centuries in the Late Holocene (Stine 1996), and recent precipitation trends in most of the assessment area have been steady or positive (Safford et al. 2012b). Inter-annual variability in precipitation is up at many stations in the assessment area, and the proportion of precipitation falling as rain vs. snow is increasing; as a result, the depth of the winter snowpack is decreasing, except in the southern assessment area, where mountain elevations are very high (Safford et al. 2012b). Recent trends in fire activity are positive, with burned area, fire size, fire frequency, and fire severity all rising in assessment area YPMC forests (Miller et al. 2009b, Miller and Safford 2012).

Cultural setting

To be completed at a later date.

Basic themes:

- The YPMC forest type in the assessment area is the most geographically extensive and the most used by modern man. Uses include recreation, timber harvest, mining, grazing, water storage, etc.
- Indigenous use by native Americans was also relatively high, especially near water sources and at lower elevations. Much of the native American use was seasonal, with more permanent habitation at lower elevations in the oak woodland belt (westside) and pinyon-juniper belt (eastside).
- Recent trends include high rates of growth in human population and housing. Sierra Nevada foothill communities are among the fastest growing in the State.
- Trends in resource extraction, especially timber harvest, have been dramatically downward. Most sawmills in the assessment area have closed, which has made forest treatment to reduce forest fuels and fire severity more difficult, both logistically and financially.
- Relatively little of YPMC forest is in wilderness areas; YPMC forest includes most of the multiple use areas in the assessment area.
- There have been major recent conflicts over land and resource use and the desire to protect remaining old forest habitat that supports a number of iconic and relatively rare carnivore species. Sierra Nevada Framework, an amendment to the National Forest resource management plans, attempted to create a platform for balancing stakeholder and ecological desires and needs across the assessment area, but it has been under litigation for the last decade.

METHODS

The “natural range of variability” (NRV) or “range of natural variation” was defined by Landres et al. (1999) as: *the ecological conditions, and the spatial and temporal variation in these conditions, that are relatively unaffected by people, within a period of time and geographical area appropriate to an expressed goal*. “Historical range of variation” (HRV) is a related concept, and was defined by Wiens et al. (2012) as: *the variation of ecological characteristics and processes*

over scales of time and space that are appropriate for a given management application. HRV was developed to allow the explicit incorporation of human influences on ecosystems into the analysis, since in most places on earth humans have been major ecological players for millennia. We evaluate human influences on assessment area ecosystems in our analyses, thus they may have been more properly called HRV assessments than NRV assessments, but Forest Service guidance for implementation of the 2012 Planning Rule adopts the term “natural range of variability”, so we have stuck with this term.

NRV/HRV assessments are tools used by managers to bring insights from historical ecology to resource management (Hayward et al. 2012). NRV/HRV characterizes variations in ecosystem function, structure and composition over scales of time and space. The basic purpose of NRV/HRV is to define the bounds of ecosystem behavior or trends in those bounds. As Morgan et al. (1994) put it: “The concept of HRV (NRV) provides a window for understanding the set of conditions and processes that sustained ecosystems prior to their recent alterations by humans.” Morgan et al. (1994), Manley et al. (1995), Landres et al. (1999), and Wiens et al. (2012) all list the purposes of conducting HRV/NRV assessments and the issues that must be considered in the assessment. These include the ecosystems of interest, the spatial and temporal scales of analysis, the ecological indicators to be assessed, whether or not to include human influences, whether to use only historical information or to use contemporary reference conditions and modeling as well, and so on.

According to Manley et al. (1995), HRV/NRV assessments should include the following steps:

1. determine key ecosystem elements (e.g., functions/processes; structures/patterns; composition),
2. identify measurable indicators for those ecosystem elements (e.g., fire frequency, tree density, species diversity), and
3. estimate values for the indicators over the selected HRV reference period.

The Appendix contains details, in outline form, of the process by which the USFS Pacific Southwest Region Ecology Program carried out these steps for the 11 ecosystem types assessed.

We are greatly indebted to the thorough HRV assessments funded by the Forest Service’s Rocky Mountain Region (Region 2) in the early 2000s (e.g., Dillon et al. 2005, Meyer et al. 2005, Veblen and Donnegan 2005). These valuable documents provided a sort of template for our efforts and set a high bar against which to gauge our own efforts.

Historical reference period

Morgan et al. (1994) and Wiens et al. (2012) note that the temporal scale of analysis will always be constrained by our ability to look clearly back through time. Certain data types permit insight into ecological patterns and processes thousands or millions of years in the past, but most reasonably decipherable data sources extend back decades, and in some cases a few centuries. In addition, climate and other environmental conditions tend to diverge more from current conditions the further one goes back into the past. As a result, most NRV/HRV assessments use the last 100-400 years as their baseline or “reference” period. In this NRV assessment, our principal

reference period was the three to four centuries before significant Euromerican settlement of the assessment area, i.e., the 16th century to the late 19th century. It is important to underline that this reference period is coincident with the Little Ice Age, and current warming trends are making future use of NRV/HRV reference conditions as management targets gradually more tenuous (this however does not diminish the value of NRV/HRV assessments, just the ways in which they are used; Millar et al. 2007; Safford et al. 2012a,c). Therefore, as recommended by Manley et al. (1995) and Jackson (2012), when available we also collected and interpreted information as far back as the beginning of the Holocene Epoch (12,000 years before present). We were especially interested in patterns and processes from warmer, drier periods in the past (Xerothermic Period, Medieval droughts), since most future climate projections for the assessment area project much warmer and somewhat drier conditions (at least during the growing season) by the end of the current century.

Spatial scale

NRV/HRV analyses are focused on change over time, but variation in space must also be considered. Our historical and contemporary reference data sources are more often than not derived from specific locations or landscapes in the assessment area, but our analysis is intended to apply to the bioregion as a whole. Wherever possible we sought data that represented the variety of different geographic regions and environmental situations that are found in the assessment area. Usually though, we simply had to accept the limitations of those data we could find, and use inference and our understanding of environmental variation across the bioregion to extend those data points to the larger assessment area. We report the geographic locations of our data sources throughout the report.

Information sources

Since the Sierra Nevada Ecosystem Project report was published in 1996 there has been a veritable explosion of scientific information on the ecology and current and past status of YPMC forests in the assessment area. Although we would like more data on every indicator we assessed, our principal problem was in trying to assemble and filter all of this information into a coherent whole.

This NRV assessment is based on both historical and contemporary reference sites and information sources. Historical data are especially useful when they precede the onset of major anthropogenic disturbances and alterations that have degraded assessment area ecosystems. These sorts of data are few however, and in most cases we were forced to resort to evaluation of information sources that postdated the settlement of Euroamericans in the assessment area. We used modern-day data from reference ecosystems whenever possible. By “contemporary reference ecosystems”, we mean current day ecosystems that have suffered relatively little degradation and may serve as a more natural reference against which degraded ecosystems may be compared. There are many advantages to using contemporary reference sites in NRV/HRV analyses, including the availability of modern data on ecosystem condition, and the fact that climatic and atmospheric conditions in current reference sites and current degraded sites are more or less equivalent. Safford et al. (2012a; p 57) note that

“We use historical data principally to understand ecological events and processes that we cannot observe firsthand, but directional changes in the baseline state (climate, air, water, soil, etc.) mean that historical conditions may make poor templates for the future. To compensate, contemporary reference ecosystems that are functioning as we desire should form part of the package of information that underlies restoration and resource management.”

The problem is that human alteration and degradation of assessment area ecosystems is so pervasive that identification of appropriate reference ecosystems is difficult in all instances, and impossible in some.

In our assessment we used direct data analysis and interpretation whenever possible, and we resorted to inference where necessary and justifiable. This NRV assessment includes comparisons to current conditions, as well as a summary of the literature regarding possible future trends, whenever that literature existed. Our focus was on peer reviewed publications, including papers in press or soon to be in press; government publications; Forest Service and other federal and state agency data; and in some cases academic theses or dissertations. Because information on the historical state of some ecosystems and ecological processes and patterns is scarce, in some cases we also refer to published anecdotal information from the mid-19th to early 20th centuries. We do not refer to anecdotal information from more recent times.

The Forest Service’s Forest Inventory and Analysis (FIA) program is the US national forest inventory. Plots are found across the US and are located randomly within a grid defined by latitude and longitude. We used a compilation of the most recent FIA data in the assessment area provided by the Pacific Southwest Region Remote Sensing Laboratory to provide current day data on many forest structure and composition variables. It is important to stress that FIA is a statistically robust sample of all stand conditions across the assessment area, including areas with reduced tree density and cover due to disturbances or harvest.

Determination of deviation from NRV

Like the Rocky Mountain Region HRV analyses (e.g., Meyer et al. 2005), our NRV assessments are based primarily on a “range of means” approach. Variation in data can be characterized in a number of ways. For example, the entire range of variation in a dataset is captured by reporting the extremes (minimum and maximum), but these extreme values more often than not represent rare cases that do not provide a clear picture of central tendencies. Use of extreme values to bound ecosystem variation makes NRV/HRV assessment more or less impossible, as nearly all values for a variable are theoretically possible at some time and at some place on the landscape. Generation of a standard deviation or standard error (the latter is standard deviation scaled to the sample size) is the usual statistical method for reporting variation, but in NRV/HRV analyses the availability of data and sample sizes are often inadequate. A middle path is to base the assessment on an estimate of the range of means from multiple sources for a given variable. This produces a narrower, more discernible, and probably more meaningful range of variation that can be quantitatively or qualitatively compared to modern data. See Figure 1 in Meyer et al. (2005) for an excellent depiction of how NRV/HRV varies with the type of variation used and the spatial scale of analysis.

Determination of deviation from NRV was accomplished by comparing the modern range of variation for some indicator variable (ideally represented by a mean, median, and standard deviation) with the range of means for the same variable from the NRV period and/or contemporary reference sites. In practice, direct statistical comparison was rarely possible, due to small sample sizes in the reference sources, the lack of measures of statistical variation, orders-of-magnitude differences in sample sizes between current and historical data when multiple historical data points did exist, or not uncommonly, the lack of concrete quantitative measures in the historical dataset. Our assessment of current deviations from NRV was necessarily deductive in nature, where we came to conclusions about the status of specific variables based to a great extent on our general knowledge about the ecosystems in question. Table 11 summarizes our conclusions about current deviation (or “departure”) from the NRV for key ecosystem elements in YPMC forests in the assessment area.

Measurement conversions

Length and height:

1 millimeter (mm) = 0.254 inches	or	100 mm = about 25 inches
1 centimeter (cm) = 2.54 inches	or	10 cm = about 25 inches
1 meter (m) = 3.28 feet	or	300 m = about 1000 feet
1 kilometer = 0.62 miles	or	100 km = about 60 miles

Area:

1 hectare (ha) = 2.47 acres	or	10 hectares = about 25 acres
1 square kilometer (sq km) = 24700 acres		
1 sq km = 0.386 square miles	or	1000 sq km = about 400 sq miles

Temperature:

1 degrees Celsius (°C) = 1.8° Fahrenheit	or	-10° C = 14° F; 0° C = 32° F; 10° C = 50°F; 20°C = 68°F, 30° C = 86° F, etc.
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NATURAL RANGE OF VARIABILITY DESCRIPTIONS

Function (including disturbance)

Extreme climatic events

Drought

NRV:

California's Mediterranean-type climate is characterized by an annual drought of 3-6 months at most stations in the assessment area (Major 1988). This is a longer drought than most stations in the northern Rocky Mountains experience in a 50 year period. Aside from the predictable annual warm season drought, California is also characterized by extremely wide variation in annual precipitation and a high dependence of annual precipitation on a small number of very wet winter storms (Dettinger et al. 2011). Most California plant taxa are therefore adapted to significant periods of time without access to atmospheric water. Nonetheless, periods of multiple, consecutive dry years can have major impacts on YPMC forests (e.g., Guarín and Taylor 2005), as evidenced by the massive die-off of conifer trees in the San Bernardino Mountains after the drought of the late 1990s and early 2000s. Drought itself is usually not the proximal cause of tree mortality however, as drought-induced stress also leads to greater susceptibility to insects and disease, and dry years tend to support more fire as well (Savage 1994, Logan et al. 2003, Fettig et al. 2007, Allen et al. 2010). Forest density can also intensify the effects of drought on tree mortality, through stress brought about by increased competition for water (Innes 1992, Dolph et al. 1995).

The Holocene Epoch, which began about 12,000 years ago, has been characterized by a fairly stable climate, but that is only in geological terms. Over the last 7000-8000 years, dry climatic periods have occurred on average every 80 to 260 years, with durations of droughts lasting 20-100 years on many occasions. Between 8000 and 6300 ybp, drought conditions dominated in the northern and eastern assessment area (during the so-called "Xerothermic Period"), and many large lakes in and near the assessment area either dried completely (e.g., Owens Lake) or dropped significantly in depth (e.g., Lake Tahoe). Aridity is thought to have been less pronounced in the southwestern assessment area (Yosemite National Park and points south) and to have eased earlier, perhaps as early as +/- 7500-7000 ybp. Over the ensuing two to three thousand years, aridity lessened and distinct wet and dry climatic phases occurred. During the last 2500 years, decades-long droughts have occurred on multiple occasions, with especially severe events occurring around 800 ybp and 650 ybp (Stine 1994, Benson et al. 2002, Mensing et al. 2004, Minnich 2007).

Biogeographically, vegetation in the assessment area responds to warmer and drier climatic periods by retreating to moister, cooler locations. For example, the alpine treeline rises during warmer periods and retreats with cooling; under extended drought, trees become more restricted to areas with permanent water. Lake Tahoe dropped below its sill during a number of Holocene droughts, and tree stumps can now be found under the lake. The Walker River between Bridgeport and Topaz Valley was dry during these droughts, and trees grew directly in the

current river channel (Stine 1994). Species composition also changes in response to drought. During dry periods in the Holocene, pollen data show that oaks, pines, junipers (*Juniperus*) and dryland shrubs (e.g., *Artemisia*, *Atriplex*) dominated much of the assessment area; less drought tolerant species like the firs and incense cedar were reduced, but then increased when conditions became cooler and/or wetter (Minnich 2007).

One of the main drivers of vegetation structure and composition is fire, and long-term shifts to drier conditions can greatly enhance fire activity. Beaty and Taylor (2009) studied Holocene fire in the Lake Tahoe Basin. They showed that maximum fire activity occurred at around 6500 ybp, at the height of the Xerothermic Period. Fire became gradually less prevalent thereafter, with notable peaks occurring in later drought periods at about 3000 and 1000-800 ybp (Beaty and Taylor 2009). Other authors have shown similar drought-fire interactions in and around the assessment area (e.g., Enfield et al. 2001, Swetnam and Baisan 2003, Whitlock et al. 2003, Trouet et al. 2010). Since the yellow pine species and black oak are more fire and drought tolerant (Table 2), drying climates and increasing fire frequency would be expected to benefit them over less tolerant tree species like the firs.

Comparison to current:

By many accounts, the late 19th and 20th centuries were anomalously wet, and have experienced few drought events compared to earlier periods of the Holocene (Hughes and Brown 1992, Haston and Michaelsen 1997). Numerous authors have argued that the climate of the recent past is likely not a reliable roadmap for the future of the western US (Saxon et al. 2005, Millar et al. 2007, Williams and Jackson 2007). Nonetheless, Allen et al. (2010) note that scientific documentation of climate-related forest mortality has been rising over the last few decades, and some recent drought episodes have been among the most severe of the last few centuries. The increasing abundance of drought intolerant species, and the very high stem densities and structural homogeneity that characterize much of the contemporary YPMC forest in the assessment area have primed these forests for more severe impacts of drought and related disturbance factors than would have likely been the case under reference conditions. A major concern is the fact that larger and older trees are often more prone to drought-induced mortality (Allen et al. 2010). In many places this may be exacerbated (or even caused) by higher densities of younger trees in the surrounding forest (Dolph et al. 1995, Ritchie et al. 2008).

Future:

Forest landscape change driven by drought tends not to be gradual, but rather episodic and rapid, since trees grow relatively slowly but die quickly (Franklin et al. 1987). Increased drought and heat have been amply documented to increase tree mortality around the globe, principally due to secondary factors like insects, disease, and fire (Adams et al. 2009, Allen et al. 2010). McDowell et al. (2008) outlined three interacting mechanisms by which drought can lead to broad-scale forest mortality:

1. Extreme drought and heat kill trees through cavitation of the xylem water column

2. Long-term water stress results in carbon deficits and metabolic limitations that result in carbon starvation and reduced capacity to defend against attack by biotic agents like insects and disease
3. Extended warm spells during droughts can drive increased populations in these biotic agents, which then overwhelm the already stressed tree hosts

Under warmer future climates, both drought-avoiding and drought-tolerating tree species may thus be negatively impacted; the former by carbon starvation, the latter by hydraulic failure (McDowell et al. 2008).

Bachelet et al. (2001) modeled future vegetation cover and biomass under a number of different Global Circulation Models (GCMs). One of their modeled response variables was a drought area index related to the Palmer Drought Severity Index. The authors found that the area of the US subjected to drought stress rose approximately linearly with increasing temperatures. The assessment area was not identified as especially sensitive to increasing drought under their scenarios, but most of their scenarios assumed increasing precipitation in California during the 21st century, which runs counter to most of the more recent GCMs (Dettinger 2005).

Liu et al. (2010) modeled the Keetch-Byram Drought Index (KBDI) for the period 2070-2100 under different GCMs. Liu et al. (2010) projected increasing drought potential across most of the contiguous US, including eastern and southern California and the assessment area. In the western US, most of the summer drying trend was predicted to depend on warming temperatures more than reductions in precipitation. They concluded that much of their study area will experience greatly increased fire potential during the course of the coming century.

Extreme precipitation events

NRV and comparison to current:

On a year to year basis, California experiences the most variable precipitation regime in the United States. In addition, no other state depends as much on a few large storms to generate precipitation: in the Mediterranean-climate part of California, a third to a half of annual precipitation falls in only five to ten wet days per year. So-called “atmospheric rivers” (ARs) generate 20-50% of the state’s precipitation totals (Dettinger et al. 2011). ARs are narrow bands (<200 km wide typically) of concentrated water vapor that develop over the oceans and direct large amounts of moisture at continental areas. Due to the influence of periodic ARs developing over the tropical Pacific, California experiences more extreme precipitation events than any other part of the US, including the hurricane-affected Gulf Coast (Dettinger et al. 2011).

The dependence of annual precipitation totals on extreme precipitation events leads to highly variable streamflows, as well as the propensity for landslides and other earth movement on unstable geologic substrates (Kerr 2006). Because they are more maritime, more tectonically active, and constituted of less consolidated bedrock, the California Coast Ranges are more heavily affected by flooding and earth movement associated with ARs, but the Sierra Nevada also experiences massive precipitation events that cause widespread flooding and landsliding. An example is the New Year’s storm of January 1997, which forced evacuations of 120,000 people,

flooded Yosemite Valley, and closed Highways 50 and 395 for months. Dettinger (2011) noted that ARs accounted for all major historical floods in some California river systems.

There is little direct evidence to compare the occurrence of ARs and other extreme precipitation events in the assessment area between current and past time periods. Paleoecological studies have documented the enhanced occurrence of extended droughts at different times of the Holocene Epoch (see Drought), and since a large proportion of California precipitation stems from AR events, we can hazard the inference that major changes in precipitation must involve changes in the frequency of ARs.

Future:

Dettinger (2011) modeled the occurrence and intensity of ARs under a variety of future climate change scenarios. He found that, under the most realistic emissions scenarios, average AR statistics did not change much, but the extremes changed markedly. For example, the frequency of larger-than-average ARs increased, the number of years with many AR episodes rose, and the temperatures of AR storms rose, which could feed into greater occurrence and magnitude of winter flooding and associated earth movement (Dettinger 2011).

Fire

Background: Fire regime

Ecological disturbances can be classified according to their characteristics, including frequency, size, season, intensity, severity, pattern, and so on. A “fire regime” describes the manner in which fires tend to occur in a given ecosystem, in a generalized sense and averaged over many fires over a long period of time. Fire regimes necessarily simplify a very complex phenomenon, but they offer a convenient and useful way to better understand and manage wildland fire (Sugihara et al. 2006). Under presettlement conditions, yellow pine and mixed conifer forests in the Sierra Nevada supported fire regimes characterized by frequent, low to moderate (or “mixed”) severity fires (Agee 1993, Arno 2000, Skinner and Taylor 2006, van Wagtendonk and Fites-Kaufman 2006, Barbour et al. 1993, 2007). These characteristics placed presettlement YPMC forests in Fire Regime I (fire return intervals 0-35 years, low to moderate severity), using the Schmidt et al. (2002) classification, which has become somewhat of a national standard. YPMC forests in the assessment area supported fire regimes limited principally by the amount of available fuels (rather than by fuel moisture), since a 3-6 month drought is typical in California’s Mediterranean-type climate and temperatures at low and moderate elevations are quite warm in the summer. As elevation increases, the role of fuel moisture becomes gradually more important, until it becomes a more important driver of fire regime than fuel load (Agee 1993, Miller and Urban 1999a). Various authors have identified the boundary between high elevation mixed conifer and red fir forests (2000-2300 m or more, depending on latitude), which is the elevation of the average freezing limit in winter storms and the zone of maximum snowfall, as a zone of major transition in fire regimes (Miller and Urban 1999a, Sugihara et al. 2006; Safford and Van de Water, in press).

Today, because of human influences since Euroamerican settlement, including timber harvest, grazing and fire suppression, many YPMC forests in the assessment area now support conditions

that are more characteristic of Fire Regimes III (FRIs 35-200 years, moderate severity) and IV (FRIs 35-200 years, high severity) (Schmidt et al. 2002). These are often referred to as “climate-limited” fire regimes, as fuel moisture, rather than fuel load, tends to drive fire occurrence and behavior (Agee 1993, Schoennagel et al. 2004). Evidence for this shift can be seen in the growing importance of climate in driving fire metrics in the assessment area over the last century (Miller and Urban 1999b, Running 2006, Morgan et al. 2008, Miller et al. 2009b).

In this section we summarize information available on the different components of the fire regime for YPMC forests before Euroamerican settlement and compare this to current conditions. At the end of the section, we summarize the results of studies that have generated qualitative or quantitative projections for future fire conditions in YPMC forests in the assessment area.

Background: The role of human ignitions prior to the 20th century

The presettlement fire record is mostly derived from fire-caused injury lesions in tree stems or charcoal in layers of sediment or peat, and we are mostly unable to discern lightning-ignited fires from anthropogenic fires. Although lightning occurrence varies temporally, lightning strike densities (LSDs) may provide a rough idea as to how the ratio of lightning to anthropogenic ignitions might have varied across the landscape (with the important caveat that LSDs will only correlate reasonably well with lightning *ignition* densities if fuels are in sufficient quantity and sufficiently dry). California is one of the least lightning-prone states in the United States, with most of the Mediterranean part of the state averaging 0 to 0.25 strikes/km²/yr (compare to the Gulf Coast, with over 6 strikes/km²/yr, or the southeastern and midwestern US, with more than 3 strikes/km²/yr [Orville 2008]). The highest LSDs in California are in the deserts of southeastern California and the eastern and higher western slopes of the Sierra Nevada, where average annual LSDs range from 0.3 to 0.55 strikes/km²/yr.

Given very low LSDs in westernmost California, pre-Euroamerican settlement fire frequencies in the California Coast Ranges were clearly due primarily to human use of fire. The relative magnitude of human inputs to the fire regime in the Sierra Nevada before Euroamerican settlement is much less certain, although anthropogenic fire was certainly a significant factor within some radius of many Native American cultural sites (Kilgore and Taylor 1979, Anderson 2005, Sugihara et al. 2006). Rightly or wrongly, some observers in the late 19th century were convinced that much of the very frequent fire that characterized presettlement YPMC forests had been set by American Indians (e.g., Manson 1906, Greeley 1907). Pyne (1982) opined that Native American management of vegetation through fire was widespread in the western US for thousands of years before Euroamerican arrival. Kilgore and Taylor (1979) noted a sharp decline in fire frequency after the 1870s in their southern Sierra Nevada study area, and correlated this with the replacement of Native American populations by Euroamerican settlers.

Fire frequencies did not follow the same late 19th century pattern everywhere, and some researchers have come to different conclusions. For example, Burcham (1960) suggested that Euroamerican-set fires between 1850 and 1900 created an artificially high frequency of fire that observers mistook as representative of Native American influences. Wagener (1961) summarized fire scar sites in the central and northern Sierra Nevada and determined that fire frequencies did not change until after 1900, when the National Forests were established. Since the arrival of Euroamericans after 1850 did not change fire frequencies in the records he studied,

Wagener (1961) concluded that most presettlement fires were probably caused by lightning, not humans. Swetnam and Baisan (2003) noted that recent data on lightning strike densities had documented much higher lightning incidence than assumed by earlier researchers. Based on their research in the southwestern US and the Sierra Nevada, they concluded that “fire regimes in large portions of these regions would probably have had similar characteristics... if people had never entered the Americas.” Parker (2002) maintained that vegetation patterns in assessment area forests when Euroamericans arrived were largely explainable based on topography, lightning ignitions, and climate. Probably the truth lies somewhere between the extreme views. Lewis (1973) noted that

“Despite a precontact population that is estimated to be among the highest in North America, (Indians) probably lacked sufficient numbers to burn all or even most of the vegetation on any regular and consistent basis, even had they so wanted.”

In the last decades of the 1800s, there was a general decrease in overall fire frequency, but an increase in large destructive fires in many parts of the Sierra Nevada, from shepherds, miners, loggers, and other forest users (Sudworth 1900, Leiberg 1902, Vankat and Major 1978, Kilgore and Taylor 1979, Turrentine et al. 1982, Barbour et al. 1993, Cermak 2005). Shepherds were especially singled out for blame in the literature (although some authors contend that shepherd burning was simply an extension of Native American practices in their absence [Vankat 1977]). Forest managers decried the loss of forest cover and timber that was accompanying Euroamerican settlement and the role that indiscriminate fire was playing (Greeley 1907, Turrentine et al. 1982). Cermak (2005) notes the plethora of photos from the period between 1880 and 1910 showing huge brush fields in and around settled or logged watersheds in the Sierra Nevada, areas that are now forest. By the end of the 19th century, much of the Sierra Nevada was being policed for fire, and by the second or third decade of the 20th century (and in some places earlier) fire frequencies began to noticeably drop as federal and state fire suppression policies and tactics were successfully implemented.

Components of the fire regime:

1. Fire frequency

NRV:

Fire frequencies are usually measured as “fire return intervals” (FRIs), which measure the number of years between fire events. The most commonly referenced measures of FRI in conifer-dominated vegetation are derived from small-scale (<4 ha) “composite” dendrochronological fire histories (which measure fire scars in the annual growth rings of trees), which include records from multiple trees in a defined area. Composite FRIs tend to represent the fire history of a given area better than point FRIs (derived from a single tree) because some fire events fail to scar every recording tree within the fire perimeter, especially in regimes characterized by frequent low intensity fire (Collins and Stephens 2007, Stephens et al. 2010, Falk et al. 2011). Composite FRIs are also more sensitive and better suited to analyzing changes in fire occurrence than point FRIs (Dieterich 1980, Swetnam and Baisan 2003). While there is some variability introduced by using composite FRIs from different sized areas, they are less likely to underestimate presettlement FRI values than point (single tree) FRIs.

Van de Water and Safford (2011) conducted an exhaustive review of the published and unpublished literature pertaining to mean, median, minimum, and maximum fire return intervals observed prior to significant Euroamerican settlement (i.e., the middle of the 19th century). Sources included fire histories derived from dendrochronological and charcoal deposition records, modeling studies, and expert quantitative estimates; a total of 298 sources were accessed. Van de Water and Safford's (2011) results for yellow pine and mixed conifer forests are given in Table 8. These values represent the NRV reference period, i.e. 16th century to about 1850. Averaged across the state, they found that mean FRIs ranged from 11-16 years in yellow pine and mixed conifer forests, and median FRIs ranged from 7-12 years. Mean minimum FRIs were around 5 years for both forest types, and mean maximum FRIs ranged from 40-80 years (Table 8; Van de Water and Safford 2011).

As Table 8 makes clear, presettlement fire frequencies were highest in the drier, lower elevation forest types (yellow pine and dry mixed conifer) and lower in moister and higher elevation stands (Caprio and Swetnam 1995, Taylor 2000, Sugihara et al. 2006, Fites-Kaufman et al. 2007, Gill and Taylor 2009). Fire frequencies are also driven directly and indirectly by local topographic variables, and various researchers have documented differences in FRIs between cool (mostly N-facing) and warm (mostly S-facing) slopes, and lower and higher slope positions and elevations. Overall, fire frequencies tended to be higher on warm slopes and upper slope positions, and lower on cool slopes and lower slope positions (Kilgore and Taylor 1979, Fites-Kaufman 1997, Taylor 2000, Beaty and Taylor 2001), but some site-to-site variability occurred and obvious topographic patterns in fire frequencies were not always apparent (Beaty and Taylor 2007, Scholl and Taylor 2010).

Through the Holocene, fire frequencies have changed significantly with major changes in climate (Millar and Woolfenden 2009). Working at Lake Tahoe, Beaty and Taylor (2009) found that fire frequencies were low in the Early Holocene, but then reached their highpoint at the height of the Middle Holocene (Xerothermic Period). Frequencies dropped gradually through the Late Holocene, with upward trends during the Medieval Warm Period. Beaty and Taylor (2009) note that current fire frequencies (under fire suppression) are at their lowest in probably the last 14000 years. Other researchers have found similar broad patterns (see citations in Beaty and Taylor 2009).

Comparison to current:

Safford and Van de Water (in press) compared pre-Euroamerican settlement FRIs to FRIs from the last century of fire records in California, using a set of fire return interval departure (FRID) metrics. Figure 5 shows one of these metrics, mean PFRID, generalized to the ecological subregions of California mapped by Miles and Goudey (1997), using only the Forest Service and Park Service lands within each subregion. "mean PFRID" represents current percent departure from the mean presettlement fire return interval (see Safford and Van de Water, in press, for details). It can be seen that most of the assessment area is highly positively departed, which means that FRIs are much longer than under presettlement conditions. Areas in Figure 5 that are greater than +33% departed have current FRIs that are at least 1.5 times longer than under presettlement conditions; areas greater than +67% departed have current FRIs that are at least 3 times longer than in presettlement times. To put this into perspective, yellow pine and dry mixed

conifer forests supported mean presettlement FRIs of about 11 years according to Van de Water and Safford (2011), which means that an average of 9.1 fires would occur over any given period of 100 years. Areas in Figure 5 that are greater than 67% departed from this presettlement FRI have experienced three fires or fewer over the last century. The map shows that the southern and eastern regions of the assessment area are less departed than the central and northern regions. Most YPMC forest types in the latter two regions are more than 85% departed, i.e. most of the landscape has seen zero to one fire over the last century. Safford and Van de Water (in press) also calculated mean PFRID by vegetation type (their Table 4). They found that yellow pine and dry mixed conifer were the most departed of 28 vegetation types analyzed (averaging 84% and 85% departure across California); moist mixed conifer was tied with aspen for third (average of 80% departure).

2. Fire rotation

NRV:

Fire rotation is the time in years required for fire to burn an area that is equivalent to the study area's total extent (Heinselman 1973, Agee 1993). Fire rotation can be thought of as a landscape-level FRI, and avoids the difficulties of extrapolating point or composite FRI data to landscapes. Mallek et al. (in review) summarized information from the published literature on presettlement fire rotations for seven major forest types in the bioregional assessment area. Results for yellow pine and mixed conifer are given in Table 9. Mean fire rotations in yellow pine and mixed conifer forests in the assessment area ranged from 22 to 31 years. As with the FRIs, presettlement fire rotations were longer in the higher and moister mixed conifer forests.

Comparison to current:

Miller et al. (2012) calculated fire rotations for the period 1984-2009 for yellow pine and mixed conifer forests in the assessment area, and compared rotations between Yosemite National Park, where fire during the study period was mostly managed rather than suppressed, and Forest Service lands in the assessment area, where most fires are suppressed. From their table 3, the area weighted average fire rotation for Forest Service-managed mixed conifer forest over the study period was about 258 years (range 95-355), for Forest Service-managed yellow pine forest it was about 280 years (range 115 to 516); the longest rotations were in the southern Cascades and Modoc Plateau, the shortest in the eastern Sierra Nevada. This compared to about 55 years fire rotation for both forest types in Yosemite National Park. In summary, YPMC fire rotations were about 10 times longer than presettlement on Forest Service lands in the assessment area, but only about 2 times longer in Yosemite (Miller et al. 2012).

3. Fire severity

Fire severity is a measure of the magnitude of the effect that fire has on an ecosystem (Sugihara et al. 2006). The severity of fire can be assessed from the standpoint of any ecosystem characteristic; here we focus on the severity of fire on the vegetation. Fire "severity" is not synonymous with fire "intensity". Fire intensity is a measure of the energy released by fire, whereas

severity results from the interactions between intensity, fire residence time, fuel and air moistures, the plant species involved, and other factors (Sugihara et al. 2006, Keeley 2009).

NRV:

In YPMC and related forest types, fires in the presettlement period were dominated by areas of low and moderate severity effects, where low flame lengths consumed shrubs, herbs, and tree regeneration but only infrequently killed adult trees (Muir 1894, Fitch 1900, Sudworth 1900, Leiberger 1902, Greeley 1907, Larsen and Woodbury 1917; Shaw and Kotok 1924, 1925, 1929). Shaw and Kotok (1925; p 2-3) concisely summarized the observations of many authors:

“Fires run through these pine forests, but comparatively lightly. Only occasionally and in the younger stands do they flare up into the tops of trees and become crown fires. For the most part the fires are confined to the underbrush and forest litter, burning fallen dead trees, and killing by heat rather than by actual flame the crowns of live, full grown trees.”

Even today, with a greatly increased component of high severity fire in YPMC forests, the proportion of fire area experiencing low and moderate severity effects is still generally larger than the proportion experiencing high severity fire (Miller and Safford 2008, 2012; Miller et al. 2009b), however the proportional balance between the three fire severity classes has changed considerably (Fig. 20).

Aside from historical accounts, estimates of past fire severity patterns in conifer forests are usually developed from studies of forest stand structure. “High severity” fires kill most trees on the landscape and leave their footprint through the mostly single-aged stands of regeneration that replace the original forest. Classic examples of vegetation types characterized by high severity fires include lodgepole pine in the northern Rocky Mountains, chaparral in southern California, and closed cone conifers like knobcone pine (*Pinus attenuata*), whose reproduction is closely linked to the passage of stand replacing fires.

The problem with estimating presettlement fire severity patterns in YPMC forests in the assessment area is that stand structures are and were extremely diverse, with stands typically composed of a mixture of young and old trees, often from a number of different species. Very few studies have found evidence of abundant large canopy gaps (>100 ha) in presettlement YPMC forests that after fire were repopulated by single-aged stands of trees (such large gaps, although still comparatively rare, are more commonly found in northern assessment area YPMC forests that supported higher densities of fir species). The prevalence of large canopy gaps caused by high severity fire is partly related to precipitation and forest productivity, as – assuming everything else equal – wetter forest types grow fuels more rapidly and support denser forest stands, and moister fuels require more heat input to combust, which results in longer fire-free intervals but higher severity fires when fire does occur. These moister forests also tend to support higher densities of shade-tolerant, relatively fire-intolerant tree species like fir, whose canopy structure is much more conducive to rapid movement of surface flames into the forest canopy. Higher elevation forests in the Sierra Nevada generally receive more precipitation and support more fir and other relatively fire-intolerant species. A similar gradient occurs from south to north, with precipitation and the importance of fire-intolerant species increasing with latitude (along the west side of the assessment area; the eastern regions are drier in general and more dominated

by drought tolerant species of pine, etc.) The elevation- and precipitation-driven increase in the natural proportion of high severity fire can be seen in the reference conditions in Figure 6, moving from left to right.

The relatively fine-grained heterogeneity that characterized forest structure in YPMC forests (see stand structure section) was the product of a largely low and moderate severity fire regime (Fire Regime I), with occasional occurrences of high severity fire in larger patches under the right weather and fuels conditions (Agee 1993, Arno 2000, Sugihara et al. 2006). It is important to remember that “low severity” fire does not imply the absence of tree mortality. Most classification systems define low severity as occurring below 25-30% canopy mortality. Taking the midpoint of that class, an average of around 12.5 to 15% of canopy trees in low severity areas would be expected to be killed by fire. “High severity” fire is usually defined as encompassing those areas where fire-caused mortality exceeds 75-80%, although recent satellite-imagery derived severity assessments (“RdNBR” assessments) usually map high severity areas in polygons where mortality is >90 or 95% (e.g., Miller et al. 2009a, 2009b, 2012). Thus, these RdNBR-based studies are mapping “stand-replacing” fire, where tree mortality is almost complete. In this assessment, we use >95% mortality as our definition of high severity fire. “Mixed severity” or “moderate severity” fire is simply the area between low and high (we use the term “moderate” in this assessment). Moderate severity areas are really areas where the intermix of low and high severity effects is at a fine enough grain to not be discernible with the analysis tools at hand. For most classifications, the midpoint of the moderate severity class falls between 50 and 60% mortality, so canopy tree loss in such areas may actually be substantial.

There are a number of ways to quantitatively estimate fire severity in presettlement YPMC forests. One is through direct evaluations of fire severity patterns in contemporary reference forests, where fire suppression does not occur or has been relaxed, and where extensive logging did not occur. Such forests occur in Yosemite National Park and in the Sierra de San Pedro Mártir of northern Baja California, Mexico. Yosemite National Park (YNP) was mostly unlogged but fire management in the park practiced strict fire suppression until the 1970s or 1980s, depending on the watershed. Since then, extensive areas of the park have been managed under a policy allowing many naturally ignited fires to burn. Collins et al. (2009) used RdNBR data to study fire severity in YNP’s Illilouette Valley, which was relaxed from strict fire suppression in the early 1970s. The watershed is dominated by moist mixed conifer, red fir, and lodgepole pine. Collins et al. (2009) did not differentiate their fire severity results by forest type, but overall they found that over a 31 year period a total of 13% of the fire areas they assessed had burned at high (stand replacing) severity.

Miller et al. (2012) analyzed RdNBR fire severity data from the entirety of Yosemite National Park, for the period 1984-2009. During their study period, about 72% of the fires they analyzed were managed for resource benefit (although five were later suppressed after they exceeded their prescriptions), while the remaining 28% were suppressed. They found that YPMC forests in the Park burned at about 14% high severity (13.7% mixed conifer, 14.6% yellow pine).

Another contemporary source of reference fire regime information for YPMC forests in the assessment area is the Sierra de San Pedro Mártir National Park in northern Baja California, Mexico. This area is in the southernmost part of the North American Mediterranean climate

zone and supports yellow pine and dry mixed conifer forests that are very similar to those of drier portions of the Sierra Nevada (Stephens and Fulé 2005). Unlike the Sierra Nevada, most of the Sierra de San Pedro Mártir was not logged and fire suppression has only been in effect for the last few decades, so the forests are much closer to pre-Euroamerican settlement conditions. Minnich et al. (2000) reported results from aerial photo interpretation of two fires that burned in the Sierra San Pedro Mártir in 1989. Using photos from 1991, they estimated that 16% of the analyzed fire area had experienced >90% mortality, thus perhaps 8% or so experienced stand replacing fire effects (>95% mortality). Aerial photo analysis is known to underestimate the area of low severity fire, as fire extent is primarily mapped based on fire effects to canopy trees, so surface fires are difficult to pick out, especially when a number of years have passed since the fire event. Minnich's analysis also took place two years after fire, whereas most RdNBR assessments occur one year after fire. Consequently, we view Minnich et al.'s (2000) numbers as an upper estimate of fire severity. Stephens et al. (2008) used field plots to measure severity in a fire area in the Sierra San Pedro Mártir. Only one of their 27 plots (4% of their sample area) experienced high severity effects (>95% mortality). We are also currently carrying out an RdNBR-based assessment of 25 years of fire severity patterns in the Sierra San Pedro Mártir; our preliminary results suggest an average of 4-7% high severity fire (>95% mortality) in YPMC forests.

Sudworth (1900) inventoried forestlands in the central Sierra Nevada (Lake Tahoe Basin to the latitude of Yosemite National Park) at the end of the 19th century. Concerning the nature of fire in his study region, he wrote (p. 558):

"The instances in this region where large timber has been killed outright by surface fires are comparatively rare. Two cases only were found... One of these burns involved less than an acre, and the other included several hundred acres. They are exceptional cases, and the killing of the trees is accounted for by the fact that long protection from fire and from all but cattle grazing had resulted in the accumulation of much fallen timber, considerable humus in depressions and on benches, and a dense undergrowth of brush and seedlings. The fires burned deep enough to badly injure the surface roots, which resulted in the subsequent death of the timber."

Leiberg (1902) carried out a field inventory of forestlands in the northern Sierra Nevada (American River watersheds to the Feather River watersheds) at the beginning of the 20th century and made estimates of the amount and severity of burning that had occurred in the region over the previous century. Euroamerican presence in the Sierra Nevada was minimal until after 1850, and exclusion of fire from most Sierra Nevada forests is not noticed in the fire scar record until at least the 1870s or 1880s (Sugihara et al. 2006), so Leiberg's (1902) results at least partly reflect presettlement conditions. Leiberg tallied burned area by watershed for the northern Sierra Nevada and estimated that 8% of the 19th century fire area had experienced "total destruction", i.e., stand replacement. Leiberg's (1902) assessment did not quantitatively discern among forest types, although most of the fires he visited had taken place in upper elevation mixed conifer and red fir forests.

Show and Kotok (1925) stated that fires in the "California pine region", which equates to YPMC forest, rarely burned the forest canopy, but killed canopy trees through heat from surface fires and successive scarring and hollowing out of the trunk, which resulted in typical fire-caused losses of about 5% of the "merchantable forest" (mature trees).

Beatty and Taylor (2001) used tree age data from forest plots and aerial photos from 1941 to reconstruct spatial variation in fire severity in the Cub Creek Research Natural Area on the Lassen National Forest. Vegetation in the area is dominated by moist mixed conifer. The earliest fires assessed for severity occurred in 1883, so Beatty and Taylor's (2001) results may not necessarily be representative of pre-Euroamerican conditions, as many destructive fires were set by settlers during the mining, logging, and grazing boom that occurred between 1850 and 1900 (Sudworth 1900, Leiberg 1902, Turrington et al. 1982, Cermak 2005). Also, severity classes were assigned based on numbers of emergent trees (i.e., how "single-aged" was the stand), so they are not easily correlated with measures of biomass loss like RdNBR. The authors found that forests in lower slope positions experienced mostly low severity fire, middle slope positions experienced mostly moderate and low severity fire, and upper slope positions experienced mostly high severity fire (Beatty and Taylor 2001). Assuming that the landscape was evenly divided among lower, middle, and higher slopes, the areal proportions amounted to about 32:30:38 (low:moderate:high).

Fire severity patterns in presettlement forests can also be modeled. The national, interagency LANDFIRE project collaborated with The Nature Conservancy to convene dozens of workshops nationwide, where state-and-transition models were built of hypothesized presettlement fire and forest successional dynamics by forest and fire ecology professionals (Long et al. 2006, Rollins and Frame 2006). Different models were developed for major vegetation types, which were known as Biophysical Settings (BpS) (available at: <http://www.landfire.gov/NationalProduct-Descriptions24.php>). The models were peer-reviewed and extensively refined and calibrated, and then delivered to the LANDFIRE modeling group, which used the BpS models to develop national fuels, vegetation, and Fire Regime Condition Class spatial data layers. In contrast to maps depicting current vegetation distributions, the LANDFIRE BpS layer is a modeled output of potential vegetation representing the distributions of vegetation types as they are hypothesized to have existed prior to Euroamerican settlement, based on topography, climate, soils, and the presettlement disturbance regime (Rollins 2009). The advantage of the LANDFIRE BpS layer, as opposed to maps of current vegetation, is that fire regimes both influence and are influenced by vegetation, so maps representing current vegetation may confound the ability to detect shifts in fire regime over time. The disadvantages of the BpS layer include the fact that the models are quantitative hypotheses and are difficult to empirically validate. Local inaccuracies in the mapped product can also make use of the map difficult at fine scales, but at regional and state-wide scales, the BpS map conforms closely to the broad biophysical gradients (e.g., Miller and Safford 2012; Safford and Van de Water, in press). In addition, the LANDFIRE BpS output is the only map of potential vegetation in California that is based on a transparent and peer-reviewed modeling process, incorporates the effects of fire, includes lands of all management jurisdictions, and extends across the entire assessment area.

Each BpS model includes documentation that summarizes, among other things, the fire severity outputs from 100 runs of 500 simulation years under presumed presettlement conditions. Mallek et al. (in review) summarized percent high severity outputs from these models and computed area-weighted averages (since multiple BpS models correspond to each of the major vegetation types analyzed in the study) for the assessment area: the yellow pine models averaged 5.9% high severity; dry mixed conifer, 7.1% high severity; moist mixed conifer, 11.6% high severity (see also Figure 6). We should note that the LANDFIRE project based their se-

verity categories on the FRCC classification, which defines high severity as >75% mortality. Using the 95% cutoff used by the RdNBR assessments and our assessment, the percent high severity values from these BpS models would be somewhat lower than those reported.

Stephens et al. (2007) provided literature- and field observation-based estimates of pre-Euroamerican fire severity in 12 forest types in California. Their measure of severity was “percent crown burned”, which correlates closely with the RdNBR-derived severity measures, as RdNBR is driven primarily by canopy loss from fire (Miller et al. 2009a). Stephens et al. (2007) used vegetation types described by Barbour and Major (1988), and estimated that an average of 5% of the tree canopy suffered burning (torching) in mixed conifer and ponderosa/shrub forest types, and 2.5% in Great Basin pine (Jeffrey pine) forests.

Quigley et al. (1996) summarized scientific work that was done to support Forest Planning in the Interior Columbia River Basin, just north of the assessment area. According to Quigley et al. (1996), presettlement burning in eastern Oregon and Washington, which support large areas of semiarid ponderosa pine-dominated forest, averaged between 10 and 15% “lethal” (high severity) fire, and 55-60% “nonlethal” (low severity) fire. Contemporary fires were very different, with annual burning averaging 30-35% lethal fire and 25-30% nonlethal fire.

Comparison to current:

Miller et al. (2009b, 2012) and Miller and Safford (2008, 2012) studied contemporary fire severity patterns on Forest Service lands in the assessment area, using RdNBR data. The time periods, the minimum sizes of fires considered and the vegetation type classification used differed somewhat among the studies. Miller and Safford (2008) and Miller et al. (2009) based their analysis on the existing forest types used in the Sierra Nevada Forest Plan Amendment, and considered severity trends between 1984 and 2004. Based on an area-weighting of their severity results, fires in yellow pine forests (ponderosa pine + eastside pine) burned about 33% of their total area at high (stand-replacing) severity, while mixed conifer forests (mixed conifer + white fir) burned at about 29% high severity. Miller et al. (2012) and Miller and Safford (2012) based their analysis on potential forest types (biophysical settings) from the LANDFIRE national mapping project, and considered severity trends between 1984 and 2009 (2010 in Miller and Safford 2012). After area-weighting their results, yellow pine types burned at about 42% high severity during the study period, and mixed conifer types at about 23% (Miller et al. 2012). Miller and Safford (2012) joined yellow pine and mixed conifer, and found the combination burned at 33% high severity during the study period. YPMC forests in Yosemite Park during the same period burned at about 14% high severity (Miller et al. 2012)

Using potential vegetation types (which are modeled from environmental and vegetation data) in comparisons between historical and current conditions avoids the complication introduced when vegetation type has changed over time, e.g. due to human management. However, it is difficult to assess the accuracy of modeled vegetation typing. Miller and Safford (2012) joined yellow pine and mixed conifer precisely due to inaccuracies they found in the vegetation classification. When we consider the combined YPMC forest type, the results using the existing vegetation mapping (~30% high severity; Miller and Safford 2008, Miller et al. 2009) are very close to those derived using the potential vegetation mapping (~33%; Miller et al. 2012, Miller and

Safford 2012). Either way, current fires in YPMC forests managed by the Forest Service in the assessment area are burning at much higher severity (30-33% high severity as an assessment area average) than was generally the case under presettlement conditions (long-term average probably <10%, ranging from 3-15%) (see Figure 6). Furthermore, since 1984 fire severity in YPMC forests has risen substantially, from an average in the low 20% in the mid 1980's to over 30% in 2010 (Figure 7). Severity in related forest types, such as black oak and white fir, has also risen over the last few decades (Miller et al. 2009).

4. High severity patch size

NRV and comparison to current:

In low and middle elevation forests, high severity patch size has also risen, with a dominance of small, scattered patches in presettlement and reference estimates, versus more contiguous coarser-grained patchiness in modern fire-suppressed forests. High severity patches more than a few hectares in size were relatively unusual (although not unknown) in fires in Sierra Nevada YPMC forests before Euroamerican settlement (Sudworth 1900, Shaw and Kotok 1924, Kilgore 1973, Stephenson 1990, Agee 1993, Skinner 1995, Skinner and Chang 1996, Weatherspoon and Skinner 1996), but in recent years high severity patches >500 ha have become a regular occurrence (Miller and Safford 2008, Miller et al. 2012). Between 1984 and 1996, mean high severity patch size in Forest Service fires in the assessment area almost doubled (Miller et al. 2009b). Comparisons between current reference yellow pine and mixed conifer forests (where full fire suppression is not practiced) and Forest Service forests (managed primarily under full fire suppression) further reflect these changes. For example, Minnich et al. (2000) reported that patches of high severity fire (>90% mortality in their study) in the Sierra San Pedro Mártir were almost entirely <16 ha in size; mean patch size was 1.7 ha. In contrast, Miller et al. (2012) found that the average size of high severity patches in modern Forest Service-managed fires in the assessment area was 12.2 ha (1.6 SE); high severity patches during the same time period in Yosemite National Park, where many fires have not been suppressed for the last four decades, averaged 4.2 ha (0.6 SE) in size. Collins and Stephens (2010) analyzed fire severity patchiness in a watershed of Yosemite National Park and found that 48% of the total high severity area was in patches >60 ha, which only comprised about 5% of the total number of patches; data from Miller and Safford (2008) show that patches >60 ha comprise 70% of high severity fire area in recent Forest Service fires.

To summarize, the NRV of high fire-severity patch size in assessment area YPMC forests was strongly dominated by a “salt-and-pepper” pattern of small areas mostly (much) less than a few hectares in size. Larger patches than a few hectares did occur, but they were rarely more than 100 ha in size. Nonetheless, such larger patches comprised perhaps half of total high severity area.

5. Fire size

NRV:

Mean fire size in California YPMC forests before settlement was much smaller than under current conditions (Figure 8). We were able to find seven published estimates (Taylor and Skinner 1998, Taylor 2000, Beaty and Taylor 2001, Taylor and Solem 2001, Collins and Stephens 2007 [which contains estimates for two separate National Park areas; raw data provided to authors]), and Scholl and Taylor 2011; and two unpublished estimates (A. Taylor, pers. comm.) of presettlement fire size statistics in YPMC-type forests in northern California. In addition, Show and Kotok (1923) reported fire sizes in assessment area and NW California forests (then, as now, most fires occurred in the YPMC belt) for the ten year period between 1911 and 1920, just as the federal government was beginning to engage in fire suppression. They found that the average size of all recorded fires was 38.3 ha (\pm 11.5 SE), and the average of all fires \geq 10 ha in size was 268.5 ha (\pm 43.8 SE). Restricting our consideration to fires \geq 10 ha to facilitate comparison, mean presettlement fire size averaged from these ten sources is 218 ha (\pm 32.8 SE, range of means = 85-457 ha). Finally, there are two published fire size estimates from contemporary reference areas (again, we only include fires \geq 10 ha in size to permit comparison among studies): mean fire size in the Sierra San Pedro Mártir, Baja California, Mexico was measured at 221 ha (167 ha if all measured fires are included) between 1925 and 1991 (Minnich et al. 2000); mean fire size in the Illilouette Creek watershed in Yosemite National Park, where most naturally ignited fires are allowed to burn, was 456 ha (61.4 ha if all fires included) between 1973 and 2006 (B. Collins, pers. comm.). The overall average of these 12 sources is 295 ha.

Note that there is evidence that fires in areas of simple topography may have often burned areas in excess of this \pm 300 ha mean. Norman and Taylor (2003) reported evidence of recurrent landscape-scale fires in an area of subdued topography in the southern Cascade Range; data from other parts of the southern Cascades show similar patterns (C. Skinner, USFS-PSW, pers. comm.). Their interpretation was that these larger fires were driven by herbaceous fuels in and around meadow complexes, where fuel loadings increased in wet years and then burned in subsequent dry years (Norman and Taylor 2003). Clearly topography and landscape conditions (presence of previously burned areas, rock outcrops, large streams, etc.) played a significant role in limiting fire size in the assessment area.

Comparison to current:

Data from the California Fire Perimeters database (<http://frap.cdf.ca.gov/data/frapgisdata>) shows that average fire size (all fires $>$ 10 ha) over the last 25 years for fires in YPMC forests in the assessment area (all jurisdictions) is 1400 ha (\pm 55 SE) (Figure 8), about seven times larger than either the assessment area historic data sources or the modern Sierra San Pedro Mártir data.

Figure 9 provides a graphic comparison of fire size frequency distributions in the assessment area (1910-2006; data from the CA Fire Perimeters database) versus the Sierra San Pedro Mártir (1925-1991; data from Minnich et al. 2000). The climates and vegetation are similar in the two areas (the latter being drier than the western and northern portions of the former). The former has experienced significant timber harvest and a century of fire suppression, the latter was not logged, and at the time of the last fire in the Figure 14ad experienced less than 15 years of fire management. A chi-squared test shows that the distributions are significantly different, with the assessment area experiencing many more large fires, and many fewer small fires than expected (Fig. 10).

Comparing the same two datasets, fires greater than and smaller than 1600 ha contributed equally to the total area burned in the Sierra San Pedro Mártir, but large fires (>1600 ha) contributed 70% of all burned area in the assessment area.

6. Annual area burned

NRV:

A number of published studies have estimated probable presettlement annual burned area, but only a few have split their estimates by vegetation type. Stephens et al. (2007) estimated burning rates for mixed conifer and yellow pine forests across California, and found that 5-15% of the total area of yellow pine and mixed conifer forests burned per year before Euroamerican settlement, depending on the fire frequency estimate used. Using the LANDFIRE BpS estimates of forest area before Euroamerican settlement (3.65 million ha for YPMC; Mallek et al., in review), this would result in a range of about 180,000 to 550,000 ha burned per year in YPMC forest types. North et al. (2012a) estimated fire areas for the Sierra Nevada *sensu stricto*, which does not include lands north of the N. Fork of the Feather River or east of the eastern Sierra Nevada foothills. Their “active management” forest types, which essentially conform to yellow pine and mixed conifer plus some hardwood forests, were estimated to experience between 77,000 and 203,000 ha of burning per year, or 5-18% of the total land area in the YPMC forest types (North et al. 2012a). Mallek et al. (in review), carried out an assessment of fire area for the same study area as the bioregional assessment, basing their calculations on fire rotation data found in the scientific literature. Mallek et al. (in review) found that presettlement burning probably affected about 4% of the land area in the YPMC forest types in an average year, or around 145,000 ha. Stephens et al. (2007) suggested that their low-end estimates of burning were probably more trustworthy, so it seems reasonable to suggest that somewhere around 5% of the area of YPMC forest types probably burned in an average year (and there was likely very high variability, year to year), or somewhere between 150,000 and 175,000 ha across the assessment area.

Comparison to current:

Mallek et al. (in review) used the CA Fire Perimeter geodatabase to calculate the total area of fire within yellow pine, dry mixed conifer and moist mixed conifer forests in the assessment area for the period 1984-2009. During the period, 489,000 ha of fire are recorded in the database for these forest types, which comes to an annual average of about 18,800 ha in YPMC forests. This is between 10% and 13% of our presettlement estimate above. Data in North et al. (2012a) suggest that current wildfire in the Sierra Nevada proper (about 60% of the assessment area) accounts for 10-30% of presettlement fire area.

7. Fire season

NRV and comparison to current

The concept of “fire season” refers qualitatively to the average period of the year during which wildfire activity can be expected. Qualitatively or quantitatively, there are a variety of ways to define the fire season. For example, Westerling et al. (2006) defined fire season as the number

of days between the first wildfire occurrence of the season and the final wildfire control date. Thonicke et al. (2001) defined the annual length of the fire season by summing the probability of at least one fire in a day across the calendar year.

The assessment area experiences one of the longest fire seasons in North America, because of the combination of fuels fed by winter and spring moisture and the 3 to 6 month period of drought that coincides with the warm season (Sugihara et al. 2006, Keeley et al. 2012). Years of widespread burning in the assessment area are driven strongly by drier than average years, and this pattern stretches back to the beginning of the fire scar record (Trouet et al. 2010).

The fire season is driven to a great extent by temporal patterns in fuel moisture, which are in turn largely dependent on winter and spring precipitation. YPMC forests in the assessment area receive from about 25-50% of their precipitation as snow (Minnich 2007), depending on elevation and latitude, among other things. Fire season in YPMC forests north of the Tehachapi Mountains (i.e., north of approximately latitude 35° N) is generally considered to run from late June to October (Fried et al. 2004, Sugihara et al. 2006), but the average length varies with latitude, elevation, topography, and of course annual precipitation and temperature. Fire season tracks and slightly lags the climatic dry season. An approximation of the fire season length and timing can be had from the Walter climate diagrams in Figure 2. Because of fuel and soil moisture carryover from the wet season, fine forest fuels are typically dry enough to burn for some time after the beginning of the climatic dry season, and coarser fuels remain dry enough to burn for some time after the beginning of the wet season (see Figure 2); a month lag on either end may be a reasonable approximation. The fire scar record shows that fires historically tended to occur late in the fire season (after cessation of summer tree growth) in the northern assessment area, but more in the middle of the fire season in the southern assessment area; areas further south continue the earlier trend, with fires in the Baja California YPMC forests tending to occur in the early to middle fire season (Skinner 2002, Stephens and Collins 2004, Skinner et al. 2008).

In a study of wildfire in the western US, Westerling et al. (2006) found that fire season length had increased by over two months during the period between 1970 and 2003. They attributed the earlier start of the fire season to earlier snowmelt from higher spring and summer temperatures.

Future fire regimes

Flannigan et al. (2009) summarized the results of dozens of future fire activity projections under different climate change scenarios and concluded that most evidence pointed to increased fire occurrence and area burned. Referring to North America in general, Gedalof (2011) similarly noted that “nearly all... (modeling) efforts predict a substantial increase in wildfire activity over the next century...” Bachelet et al. (2007) concluded that “despite imprecise knowledge of future climate and human behavior, it is reasonable to conclude that fires will likely increase in the West.” Safford et al. (2012b) stated that “modeling studies specific to California expect increased fire activity to persist and possibly accelerate under most future climate scenarios.” Miller and Urban (1999b) noted that the altered water balance that results from climate change can influence fire regimes both directly, through its influences on fuel moisture, and indirectly, through its influences on forest structure, composition, and fuel loads. Miller and Urban (1999b) concluded by stating that “If GCM predictions of future climate prove true, fires could be both

more frequent and of greater spatial extent at certain sites.” Many modeling efforts have been undertaken that either focus on or include California, and we summarize these efforts below.

Fire frequency

Miller and Urban (1999b) simulated the effects of climate change on forest biomass, composition, and fire regimes across an elevational gradient in Sequoia-Kings Canyon National Parks. For their lowest two sites, which occur within YPMC forests, fire frequencies rose markedly during the first century of their climate change simulations, but then dropped over time as woody biomass was increasingly lost. By the end of their 400 year climate change simulations, woody biomass had decreased at the 1800 m site to 0 to 4 m²/ha overall, leaving little fuel to burn; under their most extreme scenario, forest fuels at this site were completely replaced by fine (grassy) fuels and fire frequency rose. At the 2200 m site, biomass loss was also high but not as extreme as at 1800 m, and fire frequencies remained similar to the baseline conditions although fire area decreased with the decrease in biomass (Miller and Urban 1999b). The 2600 m site, which is currently in red fir forest but was predicted to transition to a mixed conifer composition, experienced very large increases in fire frequency.

Fire severity and intensity

Flannigan et al. (2000) modeled the seasonal severity rating (SSR) across North America under two Global Circulation Model (GCM) scenarios for the year 2060. SSR measures the difficulty of fire control. Flannigan et al. (2000) found that SSR increased by an average of 10% under both GCM scenarios for California. Lenihan et al. (2003a,b) modeled vegetation and fire response to different GCM-based future climate scenarios for California, using the MC1 dynamic vegetation model. One of their mid-stream outputs was fireline intensity, measured as Btu/ft/sec (see Lenihan et al. 2003b). Lawler et al. (2012) summarized the fireline intensity outputs for the range of the California fisher in the southern Sierra Nevada (most of the west slope of the Sierra Nevada in the assessment area southern region), under a moderately warmer, slightly drier future climate scenario. They found that Lenihan et al.’s model projected higher fire intensity across about 35% of the fisher range, lower intensity on 15% of the area, and similar fireline intensity on 50% of the area. Fried et al. (2004) modeled fire behavior while accounting for suppression response for lands under State of California fire management. Under a conservative future climate scenario, Fried et al. (2004) found that decreased fuel moistures and increased seasonal windspeeds under climate warming led to faster burning and more intense fires. Most of their increased fire activity was due to an increase in the number of fires burning under extreme conditions. Flannigan et al. (2013) linked the Canadian Forest Fire Weather Index to three GCMs and predicted the Cumulative Severity Rating (CSR), a fire danger metric based on weather conditions, for the northern and southern hemispheres for the periods 2041-2050 and 2091-2100. They projected that severity as measured by CSR would increase by 10 to ~30% by the later time period over most of the assessment area.

Burned area

In Miller and Urban’s (1999b) model, area burned at their 1800 m and 2200 m sites rose strongly during the first century of their climate change simulations, but then decreased over time as woody biomass was gradually lost. At the lowest site, little woody biomass remained at the end

of their simulation and the abundance of grassy fuels led to a large increase in area burned. At the 2200 m site, fire area decreased as biomass was lost over time (Miller and Urban 1999b). Their red fir forest site at 2600 m experienced very large increases in area burned. Fried et al. (2004), using a relatively conservative future climate scenario, compared six-year simulations for current and future conditions. Their future simulation burned 43% more area in mixed conifer forest than the current scenario.

Lenihan et al. (2008) simulated fire and vegetation dynamics in California under three future climate change scenarios. In all three of their scenarios, the assessment area experienced some of the greatest increases in simulated area burned (20-50% increases), especially on the Modoc Plateau, at higher areas of the southern Cascades, and in middle to high elevation forests along the Sierra Nevada axis.

Spracklen et al. (2009) built regression models linking observed climate with observed wildfire area burned and used a GCM to project burned area out to 2050. They projected an increase of about 180% in annual area burned for their Pacific Northwest region, which included most of the assessment area. National Research Council (NRC 2011) modeling projected that, compared to the average of the 1950-2003 period, median annual area burned would increase by over 300% for the assessment area with a 1°C increase in average temperature. Over time, the report noted that extensive warming and wildfire could ultimately exhaust the fuel for fire in some regions, as forests were completely burned (NRC 2011). Westerling et al. (2011) modeled burned area across California under a range of future climate and development scenarios. They found that, under the most realistic future climate and emissions scenarios and compared to the average of the period 1960-1990, area burned by wildfire increased by over 100% by 2085 for most of the forested area of northern California. Middle and higher elevations forests in the assessment area were among the most severely impacted, with some future climate scenarios producing increases in burned area of more than 300%.

McKenzie et al. (2004) calculated correlations between mean summer temperature and precipitation and annual burned area for eleven western states between 1916 and 2002, and then employed regression models to project burned area into the future under two emissions/climate scenarios. They found strong relationships between their summer climate variables and fire area for all states but California and Nevada, and concluded that most of the western US was likely to experience large increases in annual area burned by wildfire in the 21st century. However, they conclude that “fire in California and Nevada appears to be relatively insensitive to summer climate, and area burned in these states may not respond strongly to changed climate.” In their study, McKenzie et al. (2004) make two errors with respect to their analysis in California. First, they neglect to account for California’s Mediterranean-type climate, which features a summer drought of 3-6 months. Second, McKenzie et al.’s (2004) analysis bins southern and northern California, which each contribute about half of California’s total burned area in an average year but which are extremely different in their fire-climate relationships. Their analysis thus buries the relatively strong relationship that exists between fire and summer climate variables (in this case, temperature) in the assessment area and other parts of northern California (Westerling et al. 2006, Miller et al. 2009b) under the southern California fire-climate relationship, which is essentially independent of summertime temperature or precipitation (Keeley 2004). In summary, changes in summer temperature and precipitation may not have strong effects on

southern California fire area, but McKenzie et al.'s (2004) predictions for the western US in general are likely to have validity for most of the assessment area.

Fire ignitions

Since the human population of California is expected to increase to more than 50 million by 2050 – and a large proportion of that increase is forecast for the Central Valley and Sierra Nevada foothills (<http://www.dof.ca.gov/research/demographic/reports/projections/view.php>) – it is reasonable to conclude that the density of human fire ignitions will also increase over time. Educational efforts can help to reduce fire ignitions and improve public safety, but more people usually means more fire (Syphard et al. 2009).

Ignitions by natural causes may also increase over the next century. Price and Rind (1994) simulated lightning distributions and frequencies under a GCM-based future climate scenario incorporating 4.2° global warming by 2100. They projected that lightning incidence could increase by up to 30% globally. Although their results did not highlight the assessment area as a global “hotspot” of increase lightning occurrence, the combination of greater lightning incidence, warmer climates, and drier fuels leads one inexorably to the conclusion that fire activity will likely rise in most semiarid areas currently supporting forest.

Fire effects on vegetation

Fire is a major driver of vegetation change in both space and time. Fire activity and behavior is tied strongly to fuel amount, fuel moisture, and weather conditions at the time of burning (Agee 1993, Sugihara et al. 2006). All of these factors are influenced by both temperature and precipitation. There is near universal agreement that the assessment area climate will become notably warmer over the next century, but projections of future precipitation patterns are much less certain (Dettinger 2005). The effects of fire on vegetation in the assessment area will depend greatly on precipitation trends, but Bachelet et al. (2007) note that in either wetter or drier conditions, forest could be notably reduced in much of the western US in a warmer future. Under drier conditions, enhanced fire frequency could favor drought-tolerant grasses, which would further enhance ecosystem flammability and reduce woody cover. Under wetter conditions, expansion of woody plants might promote more intense fires and high mortality when drought conditions occur, ultimately reducing tree biomass.

Bachelet et al. (2007) projected that most of the assessment area would see an increase in biomass consumption by fire during the 21st century, whether warming was extreme or moderate, and whether C emissions were relatively high or low. Most pixels in the assessment area experienced increases in biomass loss of 25-67% when compared to the 1961-1990 period.

Using the same vegetation dynamics model as Bachelet et al. (2007), Lenihan et al. (2008) simulated the future distribution of terrestrial ecosystems in California under three GCM-based future climate scenarios (see Figure 10). The “MC1” model used by Lenihan et al. explicitly models fire and its effects on vegetation; modeling results were driven to a notable extent by fire. Total annual area burned in California increased under all three scenarios, ranging from 9 to 15% above the 1961-1990 mean by the end of the 21st century (Lenihan et al. 2008). Fire drove grassland expansion into former shrublands and woodlands, even under the coolest and wettest future scenario; by 2099, under the warmest and driest scenario, grassland almost completely

replaced shrublands on the Sierra Nevada west slope and also expanded greatly in the California Great Basin. Broadleaf woodland and forest replaced large areas of evergreen conifer forest under all three scenarios, with fire playing an important role in the transition, especially in the relatively warmer and drier scenarios (Lenihan et al. 2008). Under the more moderate climate scenario, annual consumption of biomass by fire by the end of the 21st century across California was about 18% greater than the 1961-1990 mean. Under the two drier and warmer scenarios, simulated biomass consumption by fire was greater at first, but as woody biomass was burned, vegetation type conversions lead to large reductions in overall biomass available for burning by the end of the century.

Fire season

Although overall precipitation in much of the assessment area has been steady to rising slightly over the last ¾ century, the snow:rain ratio and snowpack depth and persistence have all been decreasing (Safford et al. 2012b). Combined with warming temperatures, these negative trends in snow amount and storage result in earlier drying of fuels. Current trends and projections of future patterns in the snow:rain ratio and snowpack persistence thus portend longer fire seasons (Mote et al. 2005, 2006; Westerling et al. 2006, Safford et al. 2012b).

Liu et al. (2010) projected future changes in “wildfire potential” by modeling the Keetch-Byram Drought Index under a number of different future Global Circulation Model-based scenarios. Under their projections, fire season becomes a couple of months longer for much of the contiguous US, including the assessment area, by the end of the 21st century. Flannigan et al. (2013) projected that fire season length would increase by more than 20 days for most of the assessment area by 2050, and all of the assessment area by 2100.

Fire suppression

The evolution of many plant species has been greatly influenced by their long-term relationships with fire (Bond and van Wilgen 1996, Sugihara et al. 2006, Keeley et al. 2012). Human-caused changes to natural fire regimes can have significant negative impacts on the diversity and composition of native plant and animal communities (Mutch 1970, Hobbs and Huenneke 1992). Because of the high ecological and evolutionary importance of frequent fire to assessment area YPMC forests, it has been argued that the *exclusion of fire* from most of the assessment area for the last century is one the most significant ecological disturbances currently at play (Barbour et al. 1993, Sugihara et al. 2006, Fites-Kaufman et al. 2007). The scientific and management literatures are overflowing with assessments of and references to the deleterious effects of fire suppression on assessment area ecosystems (summaries in Agee 1993, SNEP 1996, Barbour et al. 1993, 2007; Sugihara et al. 2006, Keeley et al. 2012, etc.). These include altered species composition and dominance patterns, increased fuels and forest density, impacts to soils and hydrological cycles and carbon sequestration, loss of important wildlife habitat, increased fire intensity and severity, decreased human safety, threats to infrastructure, and so on.

Cermak (2005) provides a detailed consideration of the development of the fire control organization and policies in California. The desire to control fire came largely as a response to the destructive burning practices of early settlers, and also the belief that frequent fires were destroying timber and reducing the capacity of the forest to regenerate. Controversies surrounding the

use of “light burning” to reduce forest fuels and protect old growth developed in the 1910s and 20s, and again in the 1950s, but in both cases proponents of fire exclusion prevailed. In 1910 Regional Forester C. Dubois directed his Forest supervisors that fire control was the top management priority in Region 5. He followed this with publication in 1914 of *Systematic Fire Protection in the California Forests*, which Cermak (2005) calls the “most influential single document in US fire control history.” It set fire control standards (forest fires were to be controlled before they reached 10 acres [4 ha]), and it described the outlines of a formal fire control organization and processes for coordinated fire planning. In 1919, Region 5 directed Forest supervisors to suppress all fires, even on neighboring private land. In 1924, the California Board of Forestry endorsed “fire exclusion” from forestlands as state policy. A policy of overnight fire control was discussed at a national Forest Service meeting in 1935, and emerged as the famous “10 a.m.” rule in May of that year.

Federal land managers were already actively working to extinguish fires when the first Forest Reserves were established in California at the end of the 19th century, but the lack of training, coordination, planning and technology meant that their ability to stop large fires was very limited (Cermak 2005). The first trained fire crews were established in the late 1920s, and the adoption of more modern techniques and technologies gradually led to increasing success in fire suppression. Consultation of the California Fire Perimeters database (<http://frap.cdf.ca.gov/data/frapgisdata>) shows a strong drop in fire frequency and annual burned area in the 1930s and 40s. This was helped by a series of wet years in the late 1930s, but by the end of the 1940s a number of innovations had markedly improved fire fighting success, including the deployment of tanker trucks and bulldozers, the institution of “hotshot” fire crews and “smokejumpers”, and the expanded use of planes and helicopters in patrols and aerial water drops (Cermak 2005).

Patterns in fire frequency show remarkable success in fire control through most of the rest of the 20th century (the Forest Service succeeds in extinguishing 98% of all ignitions before they reach 300 acres [120 ha; Calkin et al. 2005]), but beginning in the 1980s the area of forest burned began to climb. By the 1990s, 10-yr running averages for annual burned area and average fire size were at their highest points since formal record keeping began in 1908, and the upward trend continues (Calkin et al. 2005; see figures in Miller et al. 2009b). Since 1910, twelve fires have exceeded 20,000 ha (~50,000 ac) in size in the assessment area, but eleven of these have occurred since the late 1970s, and ten of them since 1995.

Part of the trend in area burned and fire size is due to changed federal fire management policies. In the face of research and management reviews showing the detrimental ecological effects of fire exclusion on western forest ecosystems (e.g., Biswell 1961, Leopold 1963), the National Park Service began to permit prescribed fires in California the late 1960s and early 1970s, and some wilderness areas experimented with management, rather than suppression, of naturally ignited fires at the same time (Stephens and Ruth 2005). The Forest Service changed its policy from strict fire control to fire management in 1974, and formally abandoned the 10 a.m. rule in 1978 (Pyne 1982). By limiting direct attack on difficult fires, and taking greater advantage of topography, natural barriers, and weather to “indirectly” control fire size, fire management agencies themselves have played a role in the growth of large fires since the late 1970s.

Nonetheless, the evidence is overwhelming that accumulated fuels and changes in forest structure from a century of fire suppression have led to major ecosystem changes in forest types that experienced frequent, primarily low severity fires before Euroamerican settlement (e.g., Leopold et al. 1963, Parsons and DeBenedetti 1979, Agee 1993, Barbour et al. 1993, 2007; SNEP 1996, Sugihara et al. 2006, etc.). In interaction with climate warming, these forest changes are now resulting in larger and more severe fires throughout the YPMC forest belt, not only in the assessment area but across the southwestern US (Skinner and Chang 1996, Miller et al. 2009b, Dillon et al. 2011, Miller and Safford 2012). In summary, fire suppression is a major disturbance factor in assessment area YPMC forests, both in its direct modification of ecosystem composition, structure, and function, and in its contribution to increased forest fuels amounts and continuity, which are leading to deleterious effects when forest fires escape control. Nearly every other section in this chapter contains additional information pertaining to the negative ecological effects of fire suppression on YPMC forests.

Grazing

American Indian inhabitants of the assessment area did not herd animals, and livestock grazing only occurred after Euroamerican settlement. A short summary of the grazing history of YPMC forests is offered here to provide context to current conditions and to the early observations that Euroamericans made of the assessment area. For additional grazing-related information, see Forest Structure:Forest Understory:Grass and Forbs.

Appreciable livestock grazing began in assessment area YPMC forests after the arrival of American settlers after 1849. By the 1860s valley and foothill ranchers were using public lands in the Sierra Nevada on a seasonal basis to graze their herds of cattle (Turrentine et al. 1982). Sheep grazing was also practiced in much of the Sierra Nevada after about 1860 (McKelvey and Johnston 1992). The herding habits, huge numbers, and more general diet of sheep caused major effects on Sierra Nevada ecosystems, especially riparian and meadows, and probably affected fire regimes as well by reducing fine fuels. Leiberg (1902) viewed grazing, especially by sheep, as a “destructive agent to the forest by preventing reforestation.” Muir (1894) referred to sheep as “hoofed locusts”. Sudworth (1900) militated for stricter control of sheep grazing. Conflicts developed between cattle ranchers and shepherds and public concern with the effects of grazing, particularly by sheep, was one of the factors leading to the designation of the Forest Reserves in the 1890s and early 1900s. Shepherds also often set fire to the forest in the late summer or fall in order to clear the forest understory and improve forage, and in some cases these fires caused major damage to YPMC and red fir forests (Sudworth 1900, Leiberg 1902, Greeley 1907, Turrentine et al. 1982, Cermak 2005).

Vankat and Major (1978) noted that livestock grazing, especially by sheep, had affected most of Sequoia National Park. However, their references to specific records of overgrazing refer almost exclusively to montane meadows and high elevation forests, and they do not list grazing as a major change agent for YPMC forests. Sheep grazing in the park ceased in the early 20th century, so there has been nearly a century for park ecosystems to recover.

Brewer’s memoirs from 1861-1864 (Brewer 1930) refer to the scarcity of good grass cover in the Sierra Nevada, and lush areas of grass are highlighted where they occur. Brewer’s team traveled on horseback, so they were reliant on the availability of forage. Brewer’s time in the Sierra

Nevada predated heavy cattle or sheep grazing, although he makes mention of grazing in his memoirs. Sudworth's team also relied on pasturage, and Sudworth (1900) noted that unfenced forest land supported very "short forage" and expressed the opinion that sheep grazing had decimated herbaceous and grass cover in much of the central and southern Sierra Nevada, basing his statement on the "study of long-protected forest land in the same region" and conversations with older settlers. Most of Sudworth's unpublished notes refer to higher elevation locations however, not mixed conifer. Leiberg (1902) also primarily refers to higher elevations (red fir, principally) when discussing the deleterious effects of grazing.

The period between 1894 and 1904 was extremely dry across southern California (USDI 1951; but not as catastrophically dry in northern California, except between 1897 and 1899), and most of the oft-cited observations of deleterious impacts of heavy grazing on Sierra Nevada ecosystems are from this period (e.g., those cited in McKelvey and Johnston 1992). It is also important to note that the years in which Sudworth (1900) and Leiberg (1902) carried out their field studies coincided with the second longest recorded period of profound drought in California (as measured by the Palmer Drought Severity Index – PDSI), which included the 2nd driest year on record (1898), and the 2nd driest two-year span (1898-1899) (NOAA National Climate Center, 1895-2012 data: <http://www.ncdc.noaa.gov/temp-and-precip/time-series>). The extreme dryness of the soil and depleted herbaceous cover noted by observers during this period was ascribed by many of them entirely to sheep grazing, but the extreme climatic conditions certainly played a major role. "Old settlers" memories were of times before significant sheep grazing but also of much more abundant rainfall.

Whatever the case, heavy grazing by sheep of much of the assessment area clearly reduced understory cover and impacted soil in parts of the assessment area for many decades. It probably also reduced fire frequency in some parts of assessment area YPMC forests, by reducing the amount of fine fuel (although note that the general lack of summer precipitation in the assessment area results in much less grass cover on average than in yellow pine forests in the southwestern US; see the Forest Understory section below). Given that most heavy sheep grazing ceased before World War One, one question is to what extent the effects of this disturbance have lasted over the ensuing century. A further issue is that the institution of fire suppression and the cessation of heavy grazing happened at about the same time. This complicates our ability to discern the independent effects of the two disturbances. It also means that forests were densifying (and reducing understory light availability) just as understory plant communities were being freed from decades of heavy pasturage. This probably stalled understory recovery and possibly led to different patterns of succession than would have occurred had fire not been suppressed.

Insects and Disease

Background Information (text contributed by Sherri Smith, Regional Entomologist, FHP)

Table 10 lists major insects and diseases found in tree species of assessment area YPMC forests. A more complete listing and description of injurious insects and diseases can be found at (http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fsbdev3_046410.pdf).

A significant increase in ponderosa pine, Jeffrey pine, and sugar pine mortality in the Sierra Nevada range can often be attributed to moisture stress, high tree density, and elevated bark beetle activity. The primary bark beetles associated with ponderosa pine mortality are western pine beetle and mountain pine beetle. Mountain pine beetle also kills sugar pine. Jeffrey pine beetle is the primary killer of Jeffrey pine. Dwarf mistletoes and *Heterobasidion* root disease cause additional stress on host trees. Black stain root disease is scattered throughout the northern Sierra Nevada range and can be found in ponderosa and Jeffrey pine. White pine blister rust has been devastating to sugar pine since the disease entered northern California around 1930.

White fir mortality throughout the Sierra Nevada is typically attributed to fir engraver beetle, moisture stress, and *Heterobasidion* root disease. High tree density and true fir dwarf mistletoe also contribute towards decline in some areas. Douglas-fir tussock moth readily defoliates white fir in the Sierra Nevada range. Population cycles trend upward every 7-10 years and significant levels of tree mortality have been recorded during past outbreaks.

Insects rarely kill incense cedar. Bark beetles that attack incense cedar are not considered aggressive tree killers; however, when combined with drought stress they can cause mortality. During drought periods in some areas of the Sierra Nevada range small incense cedar are the first trees to show decline and die. *Heterobasidion* root disease and true mistletoe also weaken incense cedar.

Douglas-fir in the Sierra Nevada range is not as often or heavily impacted by insects or diseases typical of more northerly latitudes. However, Douglas-fir beetle, flatheaded fir borer and black stain root disease can be found in some Douglas-fir stands. Both insects are capable of killing trees, particularly drought-stressed ones. The detected incidence of black stain root disease in Douglas-fir in the Sierra Nevada is low.

Several insects and diseases can be found on native oaks. Typically the extent or severity of their effects are not widespread or protracted. Foliar injury can result from a variety of diseases, insects and mites. Wood boring beetles are usually restricted to dead or dying branches. Damage by these agents is normally secondary in nature, rather than the primary cause of branch and/or tree decline. *Armillaria* root disease and true mistletoe can commonly be found on oaks.

Uprooting and stem breakage of giant sequoia is not uncommon and can be a problem along roads and in recreation areas. *Heterobasidion* root disease is sometimes found infecting the roots of fallen trees. Tree killing of giant sequoias by insects or diseases is rare.

NRV and comparison to current:

We have little information on insect or disease occurrence in presettlement YPMC forests in the assessment area. Based on insect and forest ecology however, some inference can be made about probable changes over time. Fettig (2012) provides a list of the bark beetle species that cause “significant” mortality in the assessment area. Within YPMC forests, most research has been done on the beetles affecting yellow pine species, especially beetles from the genus *Dendroctonus*, as they can have major impacts on mortality rates in commercially important stands of trees.

It has been understood for some time that tree stand densities have a strong relationship to bark beetle-induced mortality. Higher density stands increase competition for resources (especially water and light) and reduce tree vigor, which makes individual trees less able to withstand insect attack. Various studies demonstrate that lower density YPMC stands are much less susceptible to bark beetle attack and subsequent mortality (Fettig et al. 2007).

In the current absence of frequent understory fire, bark beetles have become one of the principal agents of tree mortality in the assessment area (Manley et al. 2000, Fettig 2012). Under reference conditions, frequent fire would have interacted with insects and disease, as well as abiotic and biotic site conditions, to drive stand structure (Bonnicksen and Stone 1982, North et al. 2012b). Much more open and heterogeneous forest structure resulted, and – based on the strongly inverse stand density vs. bark beetle relationship – we can infer that bark beetle-caused mortality was probably lower than under current conditions.

Evidence from comparisons between contemporary forests in the assessment area and reference sites in Baja California support this inference. The Lake Tahoe Basin Watershed Assessment compared modern disease and insect incidence in old growth forest versus mid seral forests in the Lake Tahoe Basin and also versus old growth forests in the Sierra de San Pedro Mártir (SSPM) (Manley et al. 2000). Mid seral forests in the LTB generally supported more mortality (29% mortality), more pest incidence, and more dwarf mistletoe infestation than old growth stands (21% mortality; note that these values are overall standing mortality, and not annual mortality rates). Assessment was made after a five year drought (1987-1992), and compared to conditions in the SSPM, which experienced similar levels of drought. Overall mortality in the SSPM, which was not logged and has seen frequent fire throughout the 20th century, was only 12%. Savage (1997) assessed mortality in the SSPM versus mortality in YPMC forests in the Transverse Ranges of southern California. In her stands she found only 4% mortality in the SSPM forests versus 14% in the S. California sites. Her interpretation was that low densities of trees in the Mexican sites (kept low by recurrent fire) reduced soil moisture competition and thus tree stress. Stephens and Gill (2005) measured mortality before and after a major drought, and found that cumulative mortality in Jeffrey pine-mixed conifer forests in the Sierra San Pedro Mártir ranged from 2.7 to 3.6%. It is unclear why there is such a discrepancy between the Savage and Stephens and Gill mortality measures and the Manley et al. number. Manley et al. (2000) concluded that fire continues to be responsible for most stand thinning in the SSPM, but insects have replaced fire as the major thinning agent in the Lake Tahoe Basin.

One assessment area insect species for which presettlement outbreak dynamics have been documented is the Pandora moth (*Coloradia Pandora*), which is a defoliator of yellow pines in and around areas with very loose, usually pumice-based soils (which are required for pupation); such soils occur frequently on the eastern side of the assessment area. Pandora moth is generally not a mortality agent, but serious defoliation can result in significantly slowed growth among affected trees. Data are not from the assessment area itself, but rather from central Oregon in ponderosa pine forests. Pohl et al. (2006) identified six major outbreaks between 1800 and 2000 (mostly at 40-50 year intervals) and demonstrated a strong correlation between outbreaks and the occurrence of drought, which weaken trees' resistance and increases susceptibility to herbivory. It seems likely that similar drought-driven dynamics also characterize Pandora moth outbreaks in the east side of the assessment area.

Future:

Fettig (2012) noted that climate warming may benefit bark beetles in multiple ways. For example, warming may allow extra generations to complete their life cycles each year, and adult beetle emergence and flight could occur in the season and continue further into the fall. Under warming temperatures, cold-induced mortality during the winter may also decrease. Bentz et al. (2010) maintained that future thermal regimes, assuming continued warming, will be very favorable for many bark beetle species. Mountain pine beetles (*Dendroctonus ponderosae*) are likely to become especially damaging to higher elevation conifer forests. Huge warming and drought driven beetle outbreaks have recently occurred in the US and Canadian Rockies, and such events may be more common in the future. Bark beetles currently restricted to the southwesternmost US or Mexico will also likely move northward as climates warm.

Kliejunas (2011) carried out a risk assessment of the likelihood that current and projected future climate change trends would lead to increased adverse effects by eight pathogens on western US forests. Kliejunas (2011) estimated risk potential as the likelihood of increased disease damage multiplied by the potential consequences (which included both ecological and economic components) of increased disease damage. High to very high risk potentials were identified for dwarf mistletoes and *Armillaria* root disease under both drier and wetter future climate projections; both pathogens affect numerous conifer species in the assessment area. A moderate risk potential was identified for white pine blister rust (which affects sugar pine and western white pine) under both precipitation scenarios. Swiss needle cast (which affects Douglas-fir) and *Dothistroma* needle blight (which affects pines and Douglas-fir) were assigned moderate risk potentials under wetter future conditions, but low potential under drier future conditions.

Logging

Although American Indians used trees for a variety of purposes, large-scale timber harvest did not begin in the assessment area until after Euroamerican settlement. Like grazing, a short summary of the history of Euroamerican logging in the assessment area is provided here in order to provide some context to current conditions, and to the observations of forest conditions that were made by Euroamericans in the late 19th and early 20th centuries.

McKelvey and Johnston (1992) summarize the logging history of the Sierra Nevada up to about 1900. Most logging before the turn of the 19th century was done to support mining and the communities that arose to support mining. Timber was cut to build homes and commercial buildings, tunnels, mine and ore processing infrastructure, and railroad lines, and it was the fuel for heating, railroad engines and other machines, and the various types of mills used for processing ore; in some areas, a very valuable market in sugar pine shakes (for roofing or siding) also arose (McKelvey and Johnston 1992). The great majority of timber harvest both before and after 1900 occurred in yellow pine and mixed conifer forest.

According to McKelvey and Johnston (1992), most 19th century logging occurred on private lands near the mines and communities, at elevations below the eventual National Forests. Exceptions to this rule included the canyon of the South Fork American River, the Yuba River watershed, the Tahoe National Forest (which supported a high density of mines even at higher elevations and was crossed by the transcontinental rail line; Turrentine et al. 1982), and the basin of the

Truckee River including Lake Tahoe. Where cutting did extend into what would become National Forest lands, it was often selective rather than general, and focused on the most valuable trees, usually sugar pine and yellow pine (Sudworth 1900, Leiberg 1902).

Between the 1890s and 1920s, railroad lines were extended throughout the lower and middle elevation Sierra Nevada in order to allow access to timber resources that were beyond reach of animal-drawn transport. More than 80 private railroad logging companies existed at one time (Beesley 1996). Use of railroad for transport allowed for much more generalized and intensive logging to take place, and the northern assessment area was particularly affected, as well as areas of the Truckee River Basin and Lake Tahoe, the South Yuba River, and the Eldorado and Stanislaus National Forests (Turrentine et al. 1982, Laudenslayer and Darr 1990). According to Beesley (1996), by 1934 more than half of the mixed conifer forestland in the “north-central” Sierra Nevada had experienced some level of harvest, primarily of the more valuable pine species. Private lands were the focus of most of the harvesting, and by the 1940s almost all of the remaining uncut forestland in the Sierra Nevada was found on the National Forests or National Parks.

After the Second World War, demand for timber from federal lands increased dramatically, and the Forest Service greatly expanded its involvement in the timber industry. As an example, harvest on the Eldorado National Forest expanded from about 3.8 million board feet per year between 1902 and 1940, to 35.1 million board feet during the war, to over 56 million board feet per year between the end of the war and 1959 (Beesley 1996). Before the war, Forest Service lands provided about 5% of the nation’s supply of lumber. By the end of the war the total had risen to 10%, and by 1970 Forest Service lumber supplied almost 1/3 of the US need (Beesley 1996). In the Sierra Nevada, the period of the heaviest timber harvest occurred in the 1950s and 1960s. Harvest techniques were more generalized than before the war, and large areas of forest were clearcut. Since the 1960s, national legislation, regulations, changing economics and environmental concerns have acted in concert to greatly reduce the amount of logging occurring on public lands in the Sierra Nevada and elsewhere, although private lands have made up some of the difference. In the end, Barbour et al. (1993) estimate that “half the original acreage of the mixed conifer forest has been... cut at least once in the last 150 years.”

The effects of logging on forest stand structure have been sufficiently described in the literature and will only be summarized here. Historically most cutting in the assessment area has been at least partly selective, with the largest and most valuable trees from a handful of species (mostly sugar pine and the yellow pines) being targeted. Areas of clearcutting also occurred, mostly between the 1950s and 1970s. In both cases the tree canopy is opened up, increasing light incidence in the understory and often resulting in a pulse of herbaceous and then shrub growth, followed by ingrowth of mostly single-aged tree regeneration (either planted or not). In selectively cut areas, smaller and less valuable trees were retained, usually from fir species or incense cedar, which greatly increased their relative presence in the forest stand. Without further management and in the absence of fire, harvested stands, which may have been originally dominated by fire tolerant species like sugar pine or yellow pine, mature to much denser stands of more evenly aged individuals of shade tolerant/fire intolerant species, sometimes with scattered remnant overstory pines. Most of the shade intolerant/fire intolerant tree species are competitively dominant in the absence of recurrent disturbance (Burns and Honkala 1990, FEIS 2013; see Introduction). Compared to the original, pre-harvest forests, these forests tend to be

much denser, structurally much more homogeneous, more shaded, and less biodiverse (Agee 1993, Barbour et al. 1993, Fites-Kaufman et al. 2007). Fuel structure in these secondary forests is also highly altered. Fuel continuity is often greatly increased over the pre-harvest condition in both horizontal and vertical directions. Surface fuels are often increased due to accumulation of timber harvest “slash”, in addition to the accumulation of fuels due to fire exclusion. Fir species support low canopy base heights and often grow under the canopy of overstory dominant trees; should fire occur, fir species are easily ignited from surface fuels and act as “fire ladders” into the canopy of large trees that are otherwise protected by their thick bark or lack of lower branches. Modern timber harvest practices are designed to avoid or even to mitigate many of these negative effects, but many forest stands in the assessment area harvested before the last few decades are very departed from structural and compositional conditions that characterized presettlement forests.

Nutrient cycling

NRV and comparison to current:

Like most of the semiarid western US, tree growth in assessment area ecosystems is primarily limited by water availability rather than by nutrients (Major 1988, Stephenson 1990). In some situations and on some substrates significant nutrient limitation may occur however, such as on ultramafic (“serpentine”) rock types, or where water availability is sufficient to permit dense vegetation growth. Where nutrient limitation does occur, it is usually for nitrogen (N) or, much less commonly, phosphorus (P). We know of no studies of nutrient dynamics in YPMC forests that have not experienced long-term fire suppression. Since fire is such an important contributor to decomposition and nutrient flux (Wohlgemuth et al. 2006, Johnson et al. 2009), we focus on studies that (1) have investigated the nutrient dynamics of YPMC forests that have experienced fire, and (2) provide some insight into how fire suppression may have changed nutrient dynamics.

Compared to other temperate and boreal forest types, assessment area YPMC forests support very low rates of litter decomposition (Stohlgren 1988, Hart et al. 1992). This is due to the Mediterranean-type climate of the assessment area and general lack of precipitation during the growing season. Most decomposition occurs in the spring, often under snow, as temperatures begin to warm but soil moisture is still high (St. John and Rundel 1976, Stohlgren 1988, Johnson et al. 2009). N and P are immobilized in needle litter and slowly released. Because of the near absence of precipitation in the growing season, trees in upland YPMC forests do not tend to generate extensive roots in the soil O horizon, and the slow decomposition rates of litter in most of the assessment area mean that nutrients mineralized from the O horizon and not taken up by plants are often leached into runoff waters (Johnson et al. 2009). Spatiotemporal variability in nutrient pools and fluxes is very high in YPMC forests in the assessment area. Spatial and temporal variation in moisture availability (mostly from seasonal snowmelt) is a major driver of nutrient dynamics, but fire is an even greater source of variability in some types of nutrient flux. Johnson et al. (2009) noted that fire suppression has allowed large increases in the depth of the O horizon and subsequent buildups in N and P, which are released to stream waters by overland flow during rain events and may be significant contributors to the deterioration of Lake Tahoe water quality.

St. John and Rundel (1976) studied the effects of fire on nutrient dynamics in a giant sequoia-mixed conifer forest. Cations (Ca, K, Mg), P, and pH were all higher in burned plots, while N, C, and cation exchange capacity (CEC) all decreased in burned plots. St. John and Rundel (1976) noted that although the cations were made available for plant uptake by fire, they were also susceptible to loss by runoff unless they were taken up by plants or held in exchange sites, which are in low supply after fire due to the loss of much of the prefire organic layer and needle litter. The loss of N to volatilization could potentially also be balanced over time by any increase in nitrogen fixation promoted by fire, such as the postfire succession of nitrogen fixing shrubs (e.g., *Ceanothus* spp) or herbs (legumes) (Wohlgemuth et al. 2006).

Other studies have found similar results. For example, Chorover et al. (1994) studied soil solution chemistry before and after prescribed fire in YPMC forests in Sequoia National Park and documented very large increases in cation concentrations in soil water, and a drop in ammonium (a source of N) to below prefire levels within the year after fire; however, nitrate, another important source of N, rose and remained elevated for three years (Chorover et al. 1994). Stephens et al. (2004) found that prescribed fires in YPMC forests in the Lake Tahoe Basin released Ca and raised pH of both soil and nearby stream water. Oliver et al. (2012) studied the effects of a severe wildfire on stream chemistry in a YPMC forest and found that P, nitrate, and cations (measured with electrical conductivity) were all elevated after fire.

Sobota et al. (2013) recently reported on geographic patterns of anthropogenic N input in the US. Western portions of the assessment area are impacted by atmospheric deposition of N, from automobile exhaust and Central Valley use of synthetic fertilizers, especially in the central and southern sections. Central California is highlighted as one of the most heavily impacted areas by N input in the entire US. Maps in Sobota et al. (2013) indicate that the western portions of the southern assessment area are receiving from five to ten times (or more) the annual input of N that they probably received before Euroamerican settlement. In terrestrial ecosystems, high levels of N are known to increase invasion and survival of fast growing annual weeds, which compete strongly for water, light and nutrients with native plants, and in some cases have major effects on fire regimes (e.g., cheatgrass [*Bromus tectorum*]). High levels of N input can also have major effects on tree ecology, including stimulation of growth, changed root:shoot ratios, induced nutrient deficiencies or imbalances, and reduced drought tolerance (Bytnerowicz and Fenn 1996). Gaseous N can also act as a pollutant, and various forms of toxic and acidifying N compounds are formed with oxygen. Effects of N deposition on YPMC forests in the assessment area are not as extreme as in southern California. In both places most N arrives as dry deposition, and the summer drought and limited dry season stomatal conductance mean that plant utilization of gaseous N is more limited than in more humid regions (Bytnerowicz and Fenn 1996).

Future:

Current trends in the assessment area are for continued suppression of most fires, with escaped fires becoming larger and more severe over time. This is a fundamentally different situation than characterized presettlement YPMC forests, where fires were frequent and largely of lower severity and intensity. It is very difficult to generalize about how these trends may impact nutrient cycling in assessment area YPMC forests. One of the most significant outcomes of continued fire suppression is likely to be the enhancement of the nutrient pool in the forest litter layer and

its episodic release by severe fires. Further enhancement of nutrients will continue to arrive in the wind, in the form of nutrient deposition (mostly N) from human sources to the west. Higher levels of soil N will likely abet further invasion by annual invasive weeds and subsequent changes in forest understories and fire regimes.

Miesel et al. (2009) conducted an interesting study where they compared belowground soil and microbial variables in forest stands mechanically treated to favor ponderosa pine versus stands treated to favor large trees regardless of species. They found notable differences between the two treatment strategies, especially in soil organic carbon content and nitrogen availability, and pH. This work suggests that future soil nutrient status may depend to some degree on the tree species favored in forest management.

Successional processes

Surprisingly little empirical and quantitative documentation of successional patterns in YPMC forests in the assessment area has been published. Nonetheless, early observers of YPMC forests in the assessment area were already well acquainted with the different ecological tolerances and successional tendencies of the major tree species. For example, Sudworth (1900), Leiberg (1902), and Greeley (1907) all refer to the strong potential within YPMC forest for dense seedling recruitment of the shade tolerant/fire intolerant species in the absence of fire (see Introduction). The species differences referred to in the Introduction and in Tables 1 and 2 and Figures AA and BB interact with the environment and ecological disturbance processes to drive successional processes in YPMC forests.

Leiberg (1902) stated that the relative proportion of tree species in assessment area YPMC forests was changing due to timber harvest and fire. In general, he noted that the relative proportions of sugar pine and yellow pine were decreasing, as recruitment of young trees was not keeping up with their removal from the overstory by logging (one exception to this rule was the northern part of his central survey area). At the same time, he described “a uniform increase” in the proportions of incense cedar and white fir across the survey area. Overall, the YPMC forests that Leiberg surveyed had low densities of tree seedlings and saplings, due to effects of frequent fire. However, he noted that stands of YPMC forest that had escaped fire for 12-15 years were often filled with stands of saplings “so dense that a man can with difficulty force his way through” (p. 43). Sudworth (1900) also noted that “The frequent open spaces in yellow-pine forests are sooner or later covered with dense patches of young trees, but these thickets may in turn be swept off by fire.”

The rate of forest infilling in the absence of fire varies along environmental gradients. For example, studies in assessment area YPMC forests have found that seedling recruitment, survival, and growth are inversely related to elevation (Hunter and Van Doren 1982, van Mantgem et al. 2006), and topographic exposure and insolation are also major drivers of seedling survival and growth rates (CITE). Local soil conditions and topographically- and vegetationally-defined (e.g., nurse plants) microhabitats can also play a major role in seedling survival, young tree growth, and rates of forest succession and densification in the absence of frequent disturbance (Tappeiner and Helms 1971, Gomez et al. 2002). In the assessment area, dense shrub cover can have a major effect on future forest composition as well, as shade tolerant trees (e.g., white fir) are more likely to survive the decades it may take to overtop the shrub canopy (Stark 1965). An-

other major driver of seedling density and forest infilling is temporal coincidence between large seed crops and years with favorable climate (high precipitation, occurrence of summer thunder-showers, low summer temperatures, etc.; Burns and Honkala 1990).

Bonnicksen and Stone (1982) provide a summary of successional dynamics in moist mixed conifer forest (including giant sequoia). Bonnicksen and Stone (1982) popularized the notion of the “shifting mosaic” of successional stages on the landscape, where neighboring sites of the same ecological “potential” could be in dissimilar vegetation states due to different spatiotemporal processes and their rates. They stressed that the nature and rate of different successional pathways depended on abiotic and biotic conditions of the site in question. That said, a generally recognized truism is that white fir is the competitive dominant in most YPMC forests in the assessment area, and the long-term absence of fire will ultimately lead to white fir forest (or Douglas-fir at lower elevations in the north assessment area). Fires in the presettlement period were frequent and mostly low severity, but some aggregations of mature trees would nonetheless be periodically killed by fire, while others were left untouched, and in yet others the understory vegetation and tree regeneration would be destroyed by the passage of fire. Although fire was frequent, there were always tree clumps that had escaped fire for several FRIs, and thickets of shade tolerant species (white fir, incense cedar) would often develop in these places. Regeneration of species like giant sequoia and the yellow pines requires mineral soil and canopy gaps large enough to bring sunlight to the forest floor (Burns and Honkala 1990, Meyer and Safford 2011). Recruitment of these species thus required fire severe enough to kill clumps of canopy trees; sugar pine tends to favor smaller gaps (Burns and Honkala 1990, Oliver and Dolph 1992).

Overall, the general picture is one of very high potential for forest recruitment, especially by shade tolerant spp., with frequent fire and/or soil conditions maintaining the dominance of the pine (and in some places, giant sequoia) overstory and a more open forest condition (Sudworth 1900, Leiberg 1902, Kilgore and Taylor 1979, Bonnicksen and Stone 1982; North et al. 2002, 2005).

Models of YPMC forest successional dynamics

Kercher and Axelrod (1984) developed a Monte Carlo-based model of YPMC forest succession (“SILVA”) in order to better understand the effects of fire on forest dynamics in the Sierra Nevada. The SILVA model is complex, and includes more than 30 subroutines that model such phenomena as species-specific demographic rates (recruitment, growth, death, injury, etc.), stand structures, fire, and brush and litter dynamics. Kercher and Axelrod (1984) used SILVA to compare forest succession after a simulated clear-cut for 500 years at two different elevations, 1520 m (5000 ft) and 1830 m (6000 ft). The lower elevation site is at the upper reaches of YPMC forests historically dominated by ponderosa pine, the upper site is nearer the upper limits of YPMC forests in the Sierra Nevada and historically included a significant component of fir species.

The time-averaged results of Kercher and Axelrod’s (1984) lower elevation simulation are shown in Figure 11. Figure 12 shows the successional progression of the lower elevation YPMC stand through the 500 years of the SILVA simulations. Fire was modeled as a stochastic process with a mean return interval of 7 years. Ponderosa pine and black oak dominate the stand immediately after the initial stand-replacing disturbance, but black oak becomes subordinate to incense cedar and then white fir by 70-100 years and almost completely drops out of the stand by 300 years (Figure 12). After 200 years, overall basal area varies between 50 and 54 m²/ha, and the

relative dominance of species changes but the proportions of shade intolerant to shade tolerant species fluctuate around 70:30 until after 400 years, when the proportion of ponderosa drops. Simulations without fire supported much higher basal area of shade tolerant/fire intolerant species like white fir and incense cedar. The higher elevation simulation is in the “fir zone” and supported much more white fir than ponderosa pine, even under frequent fire (Kercher and Axelrod (1984).

Keane et al. (1990) developed the FIRESUM successional process model as an upgrade to SILVA and applied it to understanding successional dynamics in ponderosa pine/Douglas-fir forests in the inland Northwest under different fire regimes. Keane et al.’s (1990) study area includes the northern tip of the assessment area, and four of the forest types (“fire groups”) modeled by FIRESUM either occur in assessment area YPMC forests or are similar (warm, dry ponderosa pine; Grand fir [*Abies grandis* is ecologically similar to white fir]; warm, dry Douglas-fir; moist Douglas-fir). In Keane et al. (1990), FIRESUM was used to carry out a 200 year model of successional dynamics in a semiarid ponderosa pine dominated stand beginning in 1900. The major findings were that Douglas-fir was only able to establish in the stand when fire return intervals reached 50 years, but ponderosa pine still dominated the site under these conditions, with about 50% of the basal area at year 200 (*Larix occidentalis* and Douglas-fir comprised the remainder). Under no fire, Douglas-fir comprised 1/3 of total basal area by year 100, and dominated the stand by 130 years; at the end of the simulation, Douglas-fir was about 65% of the total basal area (65 m²/ha). Under the frequent fire scenarios (<50 year FRIs), most basal area was contributed by large trees, but at FRIs of 50 years and above, fuels accumulated and fires were intense, which resulted in stands of small to intermediate trees at high densities.

Both simulation studies made clear the initial advantage that the yellow pine species have in the frequent fire scenarios. Seedlings and saplings of ponderosa pine and Jeffrey pine (and sugar pine) grow rapidly in high light environments (Figure 3), and as young trees they support thicker bark than their competitors (Figure 4). Both adaptations provide for higher survival under recurrent fire. Where fire is not frequent, and/or overstory cover is high, the yellow pines are ultimately outgrown by shade tolerant species.

Future:

It is unknown how future climates and conditions may affect basic successional processes in assessment area YPMC forests. If future environmental conditions differentially affect key species in YPMC forest, then successional relationships among species may change. An example is the effect of white pine blister rust on the five-needle pines, which in YPMC forests are represented by sugar pine and to a lesser extent, western white pine. Aside from these sorts of effects, it appears likely that warming temperatures, and increasing fire activity on some of the landscape but continued fire suppression on most of the landscape, will simply accelerate the sorts of successional changes we have already been witnessing for the last half century.

Tree mortality

NRV and comparison to current:

We have little historical information on tree mortality processes or patterns in assessment area YPMC forests. Peter Decker, an observant miner whose journal from his 1849-1851 trip to California has been published (Giffen 1966), noted after three years in the Sierra Nevada that “the woods in California seem like the original or first growth. How seldom to be seen a dead tree unless fired.” Greeley (1907) came to a different conclusion, but his viewpoint was certainly colored by his formal silvicultural training. Greeley decried what he saw as “high densities” of dead, broken topped, and fire-hollowed trees. Greeley’s (1907) estimate was that one to five percent of the stems in sugar and yellow pine were dead. Since this is a cumulative mortality estimate and snags remain standing for years, the actual annual mortality rates in the forests Greeley visited were probably very low.

Although periodic drought and high interannual variability in precipitation are characteristic of the California climate, assessment area YPMC forests are generally much denser today than under presettlement conditions. Higher stand densities lead to higher competition for light, nutrients, and water. In the standard scenario, “self-thinning” ensues, where younger, smaller, or weaker individuals succumb to competition and die while larger, healthier trees benefit from the release of competition (Westoby 1984, Barbour et al. 1987). Although the mortality rates of smaller trees tend to be higher, recruitment in the small size classes is also higher, and many recent studies have documented surprisingly high mortality rates of large trees in YPMC and other forests in and around the assessment area (e.g., Dolph et al. 1995, Smith et al. 2005, Walker et al. 2006, Lutz et al. 2009, van Mantgem et al. 2009, Dolanc et al. 2012, in review; Fellows and Goulden 2012). Most researchers have concluded that some combination of water stress, stand density, and insect-driven mortality are to blame.

The few data available suggest that background mortality rates (averaged over multiple years) in assessment area forests are between about 0.25 and 1.4% for fire-suppressed forests and <0.5% for contemporary reference forests with a largely intact fire regime (Ansley and Battles 1998, Maloney and Rizzo 2002, D. Schultz, pers. comm., 2004; Stephens and Gill 2005). Modern plantation studies broadly corroborate these numbers, and show large increases in annual mortality between low and high density stands. Data in Zhang et al. (2006), for example, give annual mortality rates of between 0 and 0.8% in thinned stands of less than 332 trees/ha, versus rates of 0.6 to 2.3% in stands of more than 2450 trees/ha.

van Mantgem et al. (2009) showed that mortality rates in western US forests have strongly increased over the last four to five decades. Of the three large regions compared (Pacific Northwest, western interior US, and “California”; in the latter all of van Mantgem et al.’s (2009) study sites are located within the bioregional assessment area), the assessment area experienced the highest mortality rates and the greatest rate of increase in mortality rates (averaged over many sites, the rate changed from <1% in the 1980s to about 1.6% in the early 2000s). Further, the greatest rates of increase in mortality were found in species of pine, in (formerly) frequent-fire forests, within the elevation belt occupied by YPMC forests; overall, the highest annual rate of

mortality was found in the studied species of pine, which died at an annual rate of about 2% during the early 2000s (van Mantgem et al. 2009).

Future:

Adams et al. (2009) showed that increases in temperature increased mortality in *Pinus edulis* (a species of pinyon pine) in the southwestern US via carbon starvation. When coupled with drought, and other factors like insect outbreaks, they concluded that future warming will increase background mortality rates, and also greatly increase the risk of regional die-off events. They called for similar experimental assessments of other tree species. Carnicer et al. (2011) studied tree species responses to increased temperatures in the Iberian Peninsula, which supports a similar Mediterranean climate to much of California. They assessed 16 species, mostly pines and oaks. They found that interactions between drought, temperature, forest density, insects and disease were resulting in increased defoliation and mortality among the studied species, and they noted that climate change-driven drought pressures on Mediterranean tree species were likely to lead toward increased damage and mortality in the future. Allen et al. (2010) summarized the results of about 150 worldwide studies of tree mortality, and concluded that the data were consistent with an increase in mortality due to climate warming and drought.

Wind

NRV and comparison to current:

There is little information on the effects of wind throw on YPMC species in the assessment area. YPMC stands were historically relatively open and species adapted to these conditions must have some resistance to high winds. Larsen and Woodbury (1917) noted that wind wasn't a generally important disturbance in sugar pine stands. According to maps in Peterson (2000), California and neighboring states are subject to fewer major wind events like tornados and convective wind events ("downbursts") than any other part of the contiguous United States. Very high winds are common when winter storms arrive at the Sierra Nevada crest, but these elevations support red fir and subalpine forests for the most part. It seems unlikely that wind as a disturbance agent has changed significantly between presettlement and current times.

Hilimire et al. (2012) reported on what they termed the most extensive windthrow event on record in the Sierra Nevada. An event with winds exceeding 145 kph occurred in the central Sierra Nevada in the upper San Joaquin River basin in the fall of 2011. Thousands of mature trees were downed, mostly red fir, white fir, and lodgepole pine. In some areas more than 70% of live trees were downed. Large trees were more susceptible to uprooting than small trees, and the effects were relatively evenly distributed across species. Larger snags were also more likely to uproot than small ones (Hilimire et al. 2012). This size-dependent effect has a very different impact on forest structure than fire, which preferentially kills smaller trees.

Future:

Changing weather patterns due to climate change may increase windthrow as large-scale pressure systems shift at the continental scale (Peterson 2000). However, in the assessment area,

it seems likely that windier conditions would have greater effects on fire effects to forest than directly on the forest itself. For example, Fried et al.'s (2004) model of future fire activity under fire suppression showed that increased windspeeds during the fire season were an important factor behind the increased number of modeled fire escapes. High winds during the wet season can also influence the outcome of future fires. The 1999 Megram Fire in northwest California burned primarily through a landscape full of wind-thrown trees and broken tops which resulted from a strong wind event a few years previous. This led to more severe fire effects to the soil in some places and hampered fire control efforts (C. Skinner, USFS-PSW, pers. comm.).

Structure

Forest landscape structure

NRV:

Analyzing landscape structure in assessment area YPMC forests is difficult. YPMC forests were not characterized by large, stand-replacing disturbance events, but rather by highly frequent, low and moderate severity events that did not generally leave a coarse-grained pattern on the landscape that could be easily mapped. Even after a century of fire suppression, an increase in forest density, and the vestiges of logging and increased occurrence of high severity fire, it can be challenging to carry out the sorts of classic landscape structure assessments that are routinely done in forests of the eastern United States or northern Europe. Because of that, we urge some caution in interpreting “patchiness” in assessment area YPMC forests from standard vegetation maps or model outputs that track successional/seral stages.

Although it may have been difficult to find much coarse-grained patchiness in presettlement YPMC forests, gradients of soil depth, moisture, temperature and insolation combined to drive differences in forest structure between cool and warm aspects, high and low elevations, and upper, middle and lower slopes (species changes across these gradients are treated in Forest Composition and Species Diversity). Such landscape differences in forest structure are still noticeable today (Fites 1993, North et al. 2002, 2012b; Barbour et al. 2007, Lydersen and North 2012). The major driver of this variation is water availability. See the Introduction for a summary of the effects of topography on forest structure in the assessment area. In general, areas with higher water availability (north aspects, higher elevations, lower slopes) tend to support higher forest density and cover (and other related variables) than areas with lower water availability (south and west aspects, lower elevations, upper slopes). When disturbed by, e.g., fire, more densely forested areas tend to exhibit a more coarsely-grained landscape structure than open forest areas, because fire effects are more severe where fuels are more continuous and fuel loadings higher (Agee 1993, Sugihara et al. 2006).

Soil depth and texture are the long-term products of interactions between topographically-driven water availability and vegetation, and both of these variables also play a major role in driving vegetation conditions. Aside from the few truly unproductive soil types (serpentine soils, for example), the major effect of soil on vegetation in assessment area YPMC forests is its contribution to water availability (see Introduction). Meyer et al. (2007) found that soil depth had strong influence on the basal area and canopy cover of YPMC forest at a site in the southern Sierra

Nevada, and North et al. (2012b) described soil depth as a major driver of forest structural and compositional heterogeneity at the landscape scale.

Fire is also an important driver of forest landscape structure in assessment area YPMC forests. When high severity fire occurs, it normally results in a successional process dominated in its initial stages by montane chaparral, which includes shrub species that germinate in response to fire. The boundaries between forest and chaparral stands are quite abrupt in the first decades after fire, but in the absence of further disturbance trees slowly reoccupy these sites. This process can take many decades. If the sites burn again, the process may take centuries or the chaparral may become a semi-permanent feature in the landscape (Nagel and Taylor 2005, Skinner and Taylor 2006). The denser forests typically found in areas of higher water availability are more likely to burn at high severity due to high fuel loadings and fuel continuity, and due to the enhanced presence of fire-intolerant species like white fir (Agee 1993, Sugihara et al. 2006, North et al. 2012b). Such environments are therefore more likely to support coarse-grained landscape structures than drier forests supporting higher densities of fire tolerant species.

The proportion of presettlement landscapes occupied by defined patches of chaparral was certainly variable, but there are few sources of information to base an estimate on. Bonnicksen and Stone (1982) carried out a reconstruction study in a YPMC-giant sequoia forest in Kings Canyon National Park, and estimated that about 19% of the study site in the 1890s was occupied by shrub “aggregations” (shrubfields); by 1977, the proportion had dropped to 11%. Show and Kotok (1924) reported on the area of the National Forests in northern California that supported “brushfields” in the early 1920s, which were seral chaparral stands that had resulted from (often human-caused) fires in previously forested areas. Their estimate of 11.1% of the landscape on six National Forests in the assessment area is slightly higher than the current area of montane and mixed chaparral that occurs on productive forestland on the same National Forests (8.6%). See Table 6 and Forest structure:Forest understory and nonforest vegetation:Shrubs for more detail.

Franklin and Fites-Kaufman (1996) used contemporary conditions in assessment area National Parks (which were not logged) to estimate that a maximum of 55% of assessment area conifer forests would have been in “old-growth” status before Euroamerican arrival. It is important to qualify that Franklin and Fites-Kaufman’s (1996) concept of “old growth” was not restricted to areas of high canopy cover, and they explicitly noted that much assessment area old growth would have been in areas of open canopy.

Models of landscape structure

The interagency LANDFIRE Program developed a nationwide map of potential vegetation representing the distributions of vegetation types as they are hypothesized to have existed prior to Euroamerican settlement. The vegetation types are referred to as Biophysical Settings (BpS), and are linked to state and transition models (see the Fire Severity section for more detail). State and transition models are nonequilibrium, non-linear models linking defined states (in this case, vegetation seral stages defined by tree size and canopy cover) via pathways that are driven by forces that can transition a site between states (e.g., successional processes, disturbances) (Westoby et al. 1989, Stringham et al. 2003).

Using Van de Water and Safford's (2011) crosswalk between their presettlement fire regime types and the LANDFIRE BpS types, we determined that BpS models 610270 (Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland), 610280 (Mediterranean California Mesic Mixed Conifer Forest and Woodland), and 610310 (California Montane Jeffrey Pine-Ponderosa Pine Woodland) best represented dry mixed conifer, moist mixed conifer, and yellow pine, respectively (see www.landfire.gov for BpS descriptions). One of the outputs of the BpS state and transition models is an average end-state for many runs of each model over a 500 year period. The end states are assumed to represent a snapshot of the average landscape condition during presettlement times, with the caveat that the landscape is greater than about 5000 ha (B. Keane, USFS, pers. comm.). LANDFIRE uses these reference values to compare to current conditions in order to develop a measure of departure ("Fire Regime Condition Class", FRCC). Figure 13 provides the distribution of reference values for the three models above, portrayed as percentages of the landscape occupied by each of five seral stages: early successional (after severe fire), mid successional (closed canopy and open canopy), and late successional (open and closed canopy). The early successional stage represents anything not dominated by trees >10 cm dbh, and therefore includes areas dominated by herbaceous plants, shrubs, and seedling and saplings. The "open" mid and late successional stages include areas of medium and large trees with canopy cover <50% (<40% in the yellow pine type), the "closed" mid and late successional stages include areas of medium and large trees with canopy cover >50% (>40% in yellow pine).

The LANDFIRE BpS models predict that, under the presettlement fire regime, 15-20% of the average YPMC landscape would have been in early seral stages (herbs, shrubs, seedlings/saplings), about 35% in areas dominated by trees between 10 and 53 cm dbh (5-21"), and 45-50% in areas dominated by trees >53 cm dbh (Figure 13). Furthermore, the BpS models predict that most of the landscape was under open forests of less than 50% canopy cover ("open" stages), especially in the yellow pine and dry mixed conifer types (Figure 13). Dense, older stands ("Late closed") are predicted to have occupied around 5% of the landscape in the yellow pine and dry mixed conifer types, but around 20% of the moist mixed conifer type.

The LANDFIRE BpS models are only models, but the BpS outputs are surprisingly close to other, independent estimates of presettlement forest landscape structure in assessment area YPMC forests. Compare, for example, Franklin and Fites-Kaufman's (1996) estimate of around 55% of assessment area conifer forests in "old-growth" status before Euroamerican arrival with the 45-50% of landscape in the BpS late successional classes (Figure 13). Also, Bonnicksen and Stone's (1982) estimate of about 19% chaparral cover in their 1890 landscape is very close to the landscape proportion for early seral vegetation projected by the BpS model, as is the 15-25% range in shrub cover found in studies of unlogged (and in some cases periodically burned) YPMC forests in and near the assessment area (see Forest Structure:Forest Understory:Shrubs). Overall, we urge some caution in applying the LANDFIRE outputs, as they assume that distinct seral stages can be mapped and followed through time. As noted above, this is a tenuous assumption in YPMC forests, which under reference conditions were characterized by fine-grained heterogeneity that does not lend itself well to landscape-level mapping of structure (forest densification and increased occurrence of high severity fire in modern stands makes seral stage mapping and modeling somewhat more tractable).

Comparison to current:

Overall, it can be safely generalized that the landscape structure of current assessment area YPMC forests is more coarsely-grained (characterized by larger, more defined patches) than under presettlement conditions. This is largely due to the interacting effects of timber harvest and fire suppression. Early selective cutting removed the large pine trees from many areas of YPMC forest, and with the long-term absence of fire, natural succession has led to infilling by higher densities of young, mostly fire-intolerant species. More recent clear-cutting, mostly on private lands, has increased the representation of coarse-grained and “hard-edged” patchiness in assessment area YPMC forests. Fire suppression has greatly increased forest fuel loadings and continuity and driven a species dominance shift to fire-intolerant trees, which together have led to an increase in the occurrence of large, stand-replacing fires. Like clearcutting, these types of fires also lead to a much coarser-grained, more hard-edged pattern in assessment area forests than was typical before Euroamerican settlement. As noted in the Fire Regime section above, because of human management over the last century and a half, assessment area YPMC forests have largely transitioned from Fire Regime I, characterized by frequent, low severity fires and fine-grained heterogeneity in forest and fuels structure, to Fire Regimes III and IV, which are characterized by infrequent fires with a much greater high severity component and a much coarser-grained forest structure. More information on these fire regime and forest structure transformations is found in the Fire Regime section above (see especially High Severity Patch Size), and the Forest Structure section below (e.g., Forest Gaps/Tree Clumps).

Many studies and reviews refer to the high structural homogeneity of contemporary YPMC forests (Agee 1993, Barbour et al. 1993, 2007; SNEP 1996, Sugihara et al. 2006), and the return of more heterogeneous landscape structure is currently a major management focus (North et al. 2012a,b; North 2012). The high homogeneity of fire suppressed YPMC forest landscapes in the assessment area is being increasingly broken up by large, severe wildfires, but most of the assessment area has not experienced fire in the last century (Safford and Van de Water, in press; see Figure 13). The coarse-grained landscape structure that results from these fires is also very different from the fine-grained landscape structure that characterized presettlement forests. It is important to underline that the forest “heterogeneity” referred to by North and colleagues (North et al. 2012a,b; North 2012) is a fine-grained heterogeneity driven by the interactions between high frequencies of mostly low severity fire and topographically-driven variations in water availability.

Future:

The few models that have been run suggest increased transition of forest to chaparral, but increased transition of chaparral to grassland as well, both trends being driven by increased fire activity (Lenihan et al. 2003a,b; 2008). Cole (2010) studied paleoecological data from earlier periods of rapid climate warming in the Pleistocene and suggested that current and projected future warming trends could be expected to greatly increase the amount of early seral vegetation on the landscape. McKenzie et al. (2004) noted that, given current and projected trends in climate and fire, the long-term persistence of late seral forest in much of the western US was questionable. Based on projections as well as trends already in play in southern California, it seems likely that – especially at lower elevations – some proportion of the YPMC forest belt

will transition to shrubland and grassland over the next century. It also seems likely that forest landscape structure will become gradually more coarse-grained as fire frequency and severity continue to increase and fire suppression efforts continue to lead to forest densification in the rest of the landscape.

Forest structure

General forest structure

NRV:

Early Euroamerican visitors to assessment area YPMC forests with interests in vegetation tended to focus on the large size of individual trees, and the generally open nature of the forest canopy. William Brewer was part of the California State Geological Survey of 1860-1864, and traveled across much of the state during that period. Brewer's memoirs (Brewer 1930) contain much reference to vegetation conditions. In general, Brewer was impressed by the large size of the trees in the Sierra Nevada and he noted in multiple places how they dwarfed anything he had been familiar with in the eastern United States. Where Brewer (1930) refers to forest densities in the conifer belt, he often describes "open" conditions, or conditions of "scattered" trees. For example, his description of the Crane Flat area in current day Yosemite National Park is of "open forests of enormous trees". Brewer also describes some areas of "dense" forest however, and the general impression is one of a very heterogeneous forest landscape.

John Fremont's memoirs of his 1843-1844 expedition to California (Fremont and Smucker 1856) refers often to the immense size of conifer trees his group encountered in the Sierra Nevada. He also makes mention of open groves of pines in a number of locations, but does not once refer to forests he encountered as being dense or closed. Indeed, as his group left the San Joaquin Valley southwest of Bakersfield on their return journey to Colorado, he remarks, as they reenter the pine belt, that "we found ourselves again traveling among the old orchard-like places".

Miners' journals occasionally contain interesting and useful information about forest structure in the mid 19th century. Peter Decker's journals from 1849-1851 (Giffen 1966) refer periodically to forest conditions. He writes both of thickets of trees and areas of open or regular spacing. J.G. Bruff's diary entries similarly contain short descriptions of both open and dense forest along the path of his travels, although – unlike other early observers – he more often describes the latter condition.

Clarence King worked for Brewer's team on the geological survey, and his memoirs also contain much reference to vegetation conditions (King 1871). Referring to the Sierra-Cascade axis, which stretches from California to British Columbia, King noted the transition from more open and "grove-like" forests in California, where individual trees tended to be larger, to denser forests in Oregon and Washington, where pines ceased to be a major component. King noted that the transition was almost imperceptible from close range, but very clear at the broad scale. King (1871, p 28-29) described the YPMC forest above Visalia thus:

"Passing from the glare of the open country into the dusky forest, one seems to enter a door, and ride into a vast covered hall... You are never tired of gazing down long vis-

tas, where, in stately groups, stand tall shafts of pine... Here and there are wide open spaces around which the trees group themselves in majestic ranks."

John Muir (1894; Chapter 8) wrote that nowhere in the Sierra Nevada would one find, either "on the rocky heights (or) down in the leafiest hollows", anything approaching the dense forests found in the Amazon, or the Himalaya, the Black Forest, or the Douglas-fir forests of Oregon. In a classic passage reproduced innumerable times, he noted that:

"The inviting openness of the Sierra woods is one of their most distinguishing characteristics. The trees of all the species stand more or less apart in groves, or in small, irregular groups, enabling one to find a way nearly everywhere, along sunny colonnades and through openings that have a smooth, park-like surface, strewn with brown needles and burs."

In general, both Leiberg (1902) and Sudworth (1900; Stephens and Elliot-Fisk 1998, Stephens 2000) described highly heterogeneous forest structure in the Sierra Nevada. Both surveyors referred qualitatively and quantitatively to the large size (height and diameter) of adult trees. Referring to the "middle timber belt", which occupied elevations between 2000 and 6000 feet (600 to 1800 m) and therefore corresponds approximately to the YPMC forest type, Sudworth (1900; p 515) wrote:

"As a rule the growth is continuous but rather open... there are, however, areas of considerable extent on broad benches where the forest is dense... The trees are usually of large dimensions."

Leiberg (1902, p 32), made similar statements about the YPMC forests of the northern Sierra Nevada:

"In the eastern and trans-Sierran districts... the old-growth forests... are generally open on all slopes except the northern and on tracts with much seepage... In the central district, outside the canyon areas, the forest is of moderate density and is rarely what might be called open, except in stands of very old growth. Elsewhere large quantities of white fir and Douglas-fir with oak combine to form thickset stands. On the rocky slopes of canyons and in the great gorges of the rivers the forest is always very open and scattered."

Concerning yellow pine-dominated forests, Sudworth (1900) noted that they were rarely if ever dense, and single big trees or groups of three to six trees often stood far apart, forming a clumped but open stand structure. He observed that younger forests (up to 60 years old), which would establish in the frequent open spaces in the forest, were often very dense, but that successional processes and fire would thin them greatly over time. Incense cedar was a regular associate of ponderosa pine, and was often also in such open stands, although it was also a common riparian tree.

Manson (1906) wrote that the frequent burning in YPMC forests (he expressed the opinion that the Indians annually burned everything) suppressed seedlings and as a result the forests were "mainly composed of old trees, many badly burned at the butt." From his wide travels in the Sierra Nevada, he attested that lower and middle elevation conifer forests of all types were

“devoid of middle-aged and young trees... The light fires gave open forests through which one could readily see for great distances.”

Greeley (1907) observed that the characteristic structure of YPMC forest was one of even-aged groups, usually with a core of pine. For example, he wrote that

“in deep, moist soils... and on the north slopes, one sees bunch after bunch of six or eight mature sugar pines of nearly the same size. Where... seedlings and saplings occur at all they are usually in groups under broken cover or in narrow openings in the stand. On the same sites fir and cedar crowd the sugar pine closely and bunches or large patches of these species occur in among the groups of sugar pine. Yellow pine... seeks the drier and warmer sites... Here it is also commonly found in large even-aged groups, from open bunches of mature trees to dense thickets of saplings and seedlings.”

In his forester tone, Greeley (1907) expanded on some of the unfortunate aspects of YPMC forests for silvicultural management. One of the chief unfavorable features was the “widespread over-maturity of the timber” (i.e., the general lack of regeneration and small trees), and the high density of dead, broken topped, and fire-hollowed trees. He estimated that that one to five percent of the stems in sugar and yellow pine were dead, 10-25% were “decadent” (needed to be cut immediately to realize any timber value), 30-40% were “mature” (should be cut in 10-15 years), and only the remainder was comprised of “thrifty” trees that were still vigorously growing and could be the basis for a 30-40 year cutting rotation. In fir and incense cedar the percentage of dead and decadent trees was higher yet, and in many stands 30% of the firs over 61 cm (24 inch) dbh were unmerchantable from decay.

According to Greeley (1907), other unfavorable features of the YPMC forest from the standpoint of a forester were the very open and “irregular” nature of the stand and the fundamental role of fire in reducing stand densities, and the very strong successional pressure of the shade tolerant species, principally white fir, on YPMC forest composition. Regarding the former, Greeley decried the effects of the intermittent fires that were common in YPMC forests, especially at lower elevations. These fires removed leaf litter and humus and killed young trees, while “simply scorch(ing) the butts of larger trees”. The whole effect reminded him of the pineries of the southeastern US. The consequent open canopy resulted in much wasted growing space in YPMC forests. Greeley (1907), like Sudworth (1900) and Leiberg (1902), also described how dense stands of chaparral would arise where fires burned the tree canopy, and how such areas would be lost to tree production without human intervention.

Areas of shrubs were an important component of YPMC forests, as fire stimulates germination of species of the most important shrub genera (especially *Arctostaphylos* and *Ceanothus*), and the open canopy meant that much light reached the forest floor. Even stands of sugar pine, which we tend to equate today with denser, more productive mixed conifer stands, were often very open and with considerable underbrush (Larsen and Woodbury 1916).

In summary, in comparison to today, early observers described YPMC forests that were generally more open, more heterogeneous, and more dominated by (clumps of) large trees (King 1871, Muir 1894, Brewer 1930, Sudworth 1900, Leiberg 1902, Greeley 1907, Vankat and Major 1978,

Laudenslayer and Darr 1990, Agee 1993, Sugihara et al. 2006, Barbour et al. 1993, 2007; North et al. 2012b).

Comparison to current:

The Sierra Nevada Ecosystem Project executive summary (SNEP 1996) included the following statement about post-settlement human impacts on assessment area forest structure:

“Forest Simplification. The primary impact of 150 years of forestry on middle-elevation conifer forests has been to simplify structure (including large trees, snags, woody debris of large diameter, canopies of multiple heights and closures, and complex spatial mosaics of vegetation), and presumably function, of these forests. By reducing the structural complexity of forests, by homogenizing landscape mosaics of woody debris, snags, canopy layers, tree age and size diversity, and forest gaps, species diversity has also been reduced and simplified. At low elevations along the western boundary, ponderosa pine was preferentially removed, and throughout its range, sugar pine has decreased in abundance first through selection and later by blister rust disease. Although the situation in the Sierra differs from that in forests in the Pacific Northwest, where fragmentation leaves remnant old-growth patches surrounded by large openings, functionally the Sierran forests have been fragmented to a lesser degree by simplification.”

In the sections below, we refer more specifically to components of forest structure, including tree density, tree size and size-class distribution, tree basal area and volume, canopy cover, forest gaps and tree clumps, snags and coarse woody debris, and forest understory and nonforest vegetation.

Tree density

NRV and comparison to current:

In the assessment area, the average YPMC forest stand today is much denser than during pre-settlement times. Figure 14 shows comparisons between historical reconstructions of stand conditions in the late 19th and early 20th century with current conditions at the same sites. In all cases modern densities are much higher than the earlier reconstructed densities; differences range from 80% to 600% (Figure 14). These differences may be slightly inflated by the inability of reconstruction studies to accurately account for very small trees in the historical period, especially from species that rapidly decay such as the firs.

Stephens and Gill (2005) sampled forty nine 0.1 ha plots in the Sierra San Pedro Mártir (SSPM), in Baja California and found an average of 145.3 trees/ha (+/- 10.4 SE, range 30-320). Safford (unpub. data) measured an average of 153 trees/ha in a more widespread sample of the SSPM, and 216 trees/ha in a similar assessment of Jeffrey pine forests in the Sierra Juarez, another Baja California site. These values fall well within the reconstructed values in Figure 14.

Numerous other empirical studies in the assessment area have also documented similar patterns to those portrayed in Figure 14. These include Ansley and Battles 1998, Beaty and Taylor

(2007, 2008), Collins et al. (2011), Taylor et al. (in press), and Dolanc et al. (in review). Dolanc et al. (in prep.) compared the 1930s Forest Service inventory of the central Sierra Nevada ("VTM"; Wieslander 1935) with the modern FIA inventory and found that mixed conifer forests had experienced increases over the 70 year period of about 65% and ponderosa pine had seen density increase by 40%. The VTM inventory ignored trees below 10 cm (4") dbh, so Dolanc et al. (in prep.) also removed small trees from the FIA data to allow comparison. Including the smaller trees, we hypothesize that the overall increase in density would probably be much greater, since the major response to climate and management trends through the time period has been in the smallest size classes.

The compiled FIA data from YPMC forest plots (USFS 2013) show an average modern density (all trees greater than 10 cm dbh) of 396.5 trees/ha (+/- 292 st. dev.). This is 2.5 times more dense than the average of the Figure 14 reference sites, Stephens and Gill (2005), and Safford (unpub. data): 158.7 trees/ha (+/- 84.3 st. dev.). It is important to recall that the FIA plots include logged and severely burned areas, so the difference between FIA and reference site studies is less than it would be if we restricted the comparison only to undisturbed FIA plots.

Further information on density by species is found in the Forest Composition section. Information on tree densities by size class (diameter) is provided in the Tree Size section.

Future:

Continuation of current trends will lead to further increases in forest density, primarily among small and medium sized trees. Trends in fire and in forest mortality (see below) may counteract these tendencies to some extent.

Tree size and size-class distribution

NRV and comparison to current:

Average and maximum tree sizes in YPMC forest stands appear to have been much larger in pre-settlement times. Taylor et al. (in press), working in Jeffrey pine dominated forests in the Lake Tahoe Basin, found that the average tree size (dbh) in the modern forest was only about 60% of the average in the forest in 1873. Lydersen et al. (in press) compared stand conditions on the Stanislaus National Forest in 1929 with conditions in 2008 and found that mean tree diameter had decreased by about 26%.

Miners like Peter Decker and Joseph Bruff both referred often to the large trees they encountered. In one place on the present-day Lassen National Forest, Bruff describes a stand of pines "ten feet diameter, 200 feet high, straight as arrows" (Read and Gaines 1949, p 204). For ponderosa pine, Muir (1894) suggested that the average size of "full-grown" trees on the western slope was over 200 feet (65 m) in height and from five to six feet (150-180 cm) in diameter. Sudworth (1900) was somewhat less generous, estimating the averages at 150-180 feet (45-55 m) height and 3-4 feet (90-120 cm) dbh. Sugar pine grew much larger: Muir (1894) lists 220 feet (67 m) height and 6-8 feet (180-240 cm) dbh as common measurements for full-grown trees, but Sudworth (1900) suggested only the largest of all sugar pines reached those sizes. Table 3 lists the average sizes of "full-grown" adult trees provided by Sudworth (1900) for uncut stands

in the central Sierra Nevada. These sizes are so large as to convince various investigators that Sudworth's numbers can only refer to the best growing sites in his survey area (Stephens and Elliott-Fisk 1998, Bouldin 1999, Stephens 2000).

Early General Land Office ("GLO") land surveys also provide an idea of the sizes of trees on the late 19th century landscape. At section corners and halfway between section corners (one mile apart), the original land survey teams took formal notes on "bearing" trees, which were the nearest trees >2.5" dbh in each compass quadrant that were likely to survive over the long term. Trees between 10-14" dbh (25-36 cm) were apparently preferred if they were available (Hyde 2002). The survey teams identified the species, marked the tree, and measured the diameter, bearing, and distance from the section corner to each tree. At quarter-section corners along the section lines two trees were marked and measured (Hawes 1882).

Hyde (2002) summarized GLO data from the Stanislaus, Sierra, and Sequoia National Forests. The average witness/bearing tree sampled by the GLO teams was <61 cm dbh (24"), with the average sampled oak being between 38 and 50 cm (15-20") dbh, the average sampled pine between 56 and 76 cm (22-30") dbh, and the average sampled fir between 21 and 71 cm (20-28") dbh (values estimated from Table 4.5 in Hyde 2002). On the Stanislaus National Forest the average sampled pine was over 78 cm (31") dbh, half of all pines sampled there were above 86 cm (34") dbh and more than 1/3 were greater than 102 cm (40") dbh; fir on the Stanislaus were nearly as big on average but the median was between 61 and 76 cm (24-29") dbh. Trees on the Sierra and Sequoia National Forests were 15-30% smaller in diameter on average (Hyde 2002). GLO data from the Lake Tahoe Basin give an average tree diameter of 61 cm (24") in YPMC forests (Manley et al. 2000), and unpublished GLO data from the Eldorado National Forest show average yellow pine and sugar pine diameters between 76 and 83 cm (30-33") dbh (J. Fites-Kaufman, pers. comm.)

The drop in the average size of trees in YPMC forests over the last century is the result of two trends, a great and general increase in the density of small trees, combined with a decrease in the number of large trees. Many studies have documented the former pattern (referenced throughout this assessment), but the latter may be ecologically just as significant. Figure 15 is reproduced from Dolanc et al. (in review), and shows changes in density between the 1930s and early 2000s, as documented by the Forest Service VTM and FIA inventories. Other studies documenting the loss of larger trees in assessment area YPMC forests include Smith et al. (2005), Lutz et al. (2009), and van Mantgem et al. (2009); note that Collins et al. (2011) did not find this pattern. Although timber harvest certainly explains some of this trend across the assessment area (e.g., Taylor 2004; Lydersen et al., in press), the patterns also occur in unlogged forests. Other factors might include insect and disease outbreaks, pathogens, and drought stress, probably exacerbated by the much higher stand densities that characterize modern YPMC forests.

Figure 16 shows reconstructed size class distributions combined from four study sites in the assessment area. Figure 16 is given as percent of measured trees, so as to better allow comparison of the shape of the reference distributions among sites. Figure 17 reproduces size class distributions from three modern reference sites that were not logged and have not experienced total fire suppression. In both figures the roughly flat or even hump-shaped distribution of tree sizes in the historical reconstructions can be appreciated. This is very different from the classic

“reverse J-shaped” age or size distribution that is typical of stable, self-replacing climax communities (Barbour et al. 1987, Oliver 2001). Assuming that size is roughly correlated with age, the X-axes of Figures 14-16 represent the spectrum from juveniles through immature, mature, and then senescent individuals. Given the relatively high probability of mortality for juveniles, a high number of young trees is usually necessary to replace the relatively few adults that die in any given period. This leads to the J-shaped (or “hockey stick”) size distribution in the contemporary forest stands in Figure 18. Such a distribution is typical of a forest community relatively free of disturbance (Smith et al. 2009). If a population is composed mostly of mature and senescent individuals, the population may be in decline (which is how early foresters interpreted the situation they encountered, e.g. Sudworth 1900, Greeley 1907), or it may be one that is replaced only by episodically successful recruitment (Barbour et al. 1987). This was largely the case with presettlement YPMC forests in the assessment area, where recurrent fire killed most juvenile trees and successful recruitment was somewhat of a stochastic event, when seed production happened to coincide with a period of sufficient precipitation and little or no fire. Roughly flat or hump-shaped age or size distributions are characteristic of old-growth forests in areas that experience frequent, low severity fire (Oliver 2001, North et al. 2007, Lydersen and North 2012). As a result, they are also found in stand reconstructions of presettlement ponderosa pine forest in the US southwest (Mast et al. 1999) and in forests of longleaf pine (*Pinus palustris*) in the southeastern US (Heyward 1939). The distribution of size classes in the GLO data from the Lake Tahoe Basin show a similar humped shape (Manley et al. 2000). Oliver (2001; his Figure 2) graphs the size-class distribution from the Beaver Creek Pinery, a well-known reference ponderosa pine-black oak site in the northern assessment area which was not logged and has experienced a number of fires in the 20th century; it is strongly hump-shaped. Using a project at Blacks Mountain Experimental Forest in the same general area, Oliver (2001) showed that prescribed fire could convert a reverse J-shaped distribution to a hump-shaped distribution.

Not all reference YPMC forests show the hump shaped distribution of tree size classes. In their array of plots in the contemporary Sierra San Pedro Mártir, Stephens and Gill (2005) found a J-shaped distribution of size classes.

Figure 19 shows the average size-class distribution in 20 cm classes for the FIA plot data compilation (USFS 2013). The obvious reverse J-shaped curve is very different from the flat or hump-shaped size distribution found in presettlement forests.

Future:

Continued high levels of recruitment among shade tolerant species, combined with increasing mortality among larger trees (see Tree Mortality), will likely intensify tendency toward J-shaped size-class distribution.

Tree basal area/volume (forest biomass)

NRV:

A number of stand reconstruction studies report late 19th century basal areas. Scholl and Taylor (2010) found that overall reconstructed basal area in 1899 was between 29 and 30 m²/ha,

depending on the smallest tree size considered. A USDA (1911) report referenced by Scholl and Taylor (2010) found basal areas in western Yosemite Park of about 21 m²/ha. Taylor et al. (in press) report an overall basal area of about 27 m²/ha in YPMC forests in the Lake Tahoe Basin (reconstructed date = 1893). Taylor (2004) reconstructed Jeffrey pine stands on the east shore of Lake Tahoe and found average basal areas of 25.5 m²/ha for the period between 1870 and 1900. Parsons and DeBenedetti (1979) provide data that produce basal areas (in 1875) of 38 to 89 m²/ha in yellow pine and productive mixed conifer forests in the southern Sierra Nevada. North et al. (2007) report an 1865 basal area of about 51 m²/ha for a moist mixed conifer forest on the Sierra National Forest.

Outside of the assessment area but in similar forest, Avery et al. (1976) used long-term stand records in ponderosa pine forest in Arizona to determine that basal area in 1920, just as federal fire suppression was gaining traction, averaged about 14.2 m²/ha across 16 forest inventory plots.

Data from contemporary reference sites give similar numbers. Lydersen and North (2012) found that modern old growth stands with recent recurrent fire averaged 54 m²/ha in a sample of 48 sites across much of the assessment area. Stephens and Gill (2005) measured stand characteristics in a limited area of Jeffrey pine-mixed conifer forest in the Sierra San Pedro Mártir in Baja California and found that basal area averaged 19.9 m²/ha, but with an enormous range (5.7 to 50.7 m²/ha). Safford (unpub. data) found average basal areas of 17.9 to 22.8 m²/ha for Jeffrey pine and mixed conifer stands across the plateau of the Sierra San Pedro Mártir, and 15.5 m²/ha for Jeffrey pine stands in the Sierra Juarez, which is at the lowest altitudinal limit of Jeffrey pine in Baja California. Taylor (2010) measured basal areas of 27.1 m²/ha in the Beaver Creek Pinery on the Lassen National Forest, which is an old growth ponderosa pine-black oak forest with multiple 20th century fire entries.

Outlying values are given by Sudworth (1900, and in his unpublished notes), who sampled plots in highly productive sites in the central and southern Sierra Nevada. Mean basal areas from plots in mixed conifer stands ranged from 130 to 270 m²/ha. The representativeness of Sudworth's (1900) plot data has been questioned (Stephens and Elliott-Fisk 1998, Bouldin 1999, Stephens 2000).

Comparison to current:

A compilation of the most recent FIA data in the assessment area (USFS 2013) suggests that mean basal area in YPMC forests has not changed dramatically over the last 100+ years. The average of the 13 reference values given above (not including Sudworth) is 33.2 m²/ha (+/- 20.9 st. dev), while the average of the 2508 FIA plots in YPMC forests included in the compilation is 32.9 m²/ha (+/- 20.4 st. dev.; all trees greater than 5 cm dbh). Mean basal area in YPMC FIA plots on the west side of the assessment area (north, central, and south subregions) is 34.6 m²/ha; on the east side (northeast and southeast subregions) it is 26.5 m²/ha.

Obviously, it is statistically somewhat tenuous to compare the results of 13 local studies with an average derived from 2508 plots regularly spaced across the entire assessment area, especially when the latter comprise all stand conditions, including logged and severely burned areas. Studies that have directly reconstructed early stand conditions on a site and then compared them to the current forest stand have mostly found that basal area has increased, although not at the

rate of tree density (see above). Figure 20 summarizes temporal changes in basal area by species for four sites in assessment area YPMC forests. Overall increases in basal area from these studies and others referenced above range from 10 to 150%, averaging about 60% across studies.

Ansley and Battles (1998) compared modern YPMC forest structure to structure in a stand first sampled 40 years before. They found that basal area rose by 15% over the 40 years, and it increased for all species except for sugar pine (which remained the same) and ponderosa pine (which dropped). Lydersen et al. (in press) compared forest stand structure conditions in 1889 (estimated through reconstruction), 1929, and 2008 on the Stanislaus-Tuolumne Experimental Forest and found that basal area had increased from 52.8 to 56.6 to 70.5 m²/ha. Increases in basal area were primarily in shade tolerant species; the percentage of pine in the sampled stands dropped from 25% to 11% over the same period.

Because presettlement YPMC forests in the assessment area generally supported more large trees than current forests, a rise in basal area over time would be driven mostly by increases in small and medium tree densities. In places where those young tree densities have not risen as rapidly, or where large tree densities have not dropped, basal areas of current forest may be similar to or even lower than presettlement forests (North et al. 2007).

Interestingly, comparisons between current FIA data and the Forest Service VTM dataset from the 1930s have mostly found that basal area and volume have *decreased* over time (e.g., Bouldin 1999, Fellows and Goulden 2008). It is difficult to determine why this is. There is some concern that VTM plots may have been preferentially sited in areas of better growing conditions or larger trees (Bouldin 2009), whereas the FIA data are a statistically unbiased sample. Little note is made in the VTM plot notes of previous logging, so it may be that they tended to avoid previously harvested stands. This would provide maybe a better idea of natural forest conditions in the early 20th century, but it would complicate comparisons with modern inventories, which include all stand conditions to permit a statistically robust estimate of landscape-level conditions. Another issue with the VTM plots is that they did not sample any trees <10 cm dbh (4"). The general understanding is that presettlement YPMC forests generally supported very low densities of saplings and small trees due to frequent fire. The VTM plots were sampled a few decades after the institution of fire suppression, so it seems likely that the absence of the smallest tree sizes in the VTM data does not constitute a major omission when calculating biomass. However, modern YPMC forests are highly enriched in the smallest size classes, and high densities of <10 cm dbh trees can contribute surprising amounts of biomass to a forest stand. Modern comparisons of, e.g., FIA plots with VTM plots have thus far removed trees <10 cm dbh from the modern plots to permit an unbiased comparison (e.g., Fellows and Goulden 2008), but this removal may result in modern plots appearing to support less biomass than the early plots when they actually don't.

In summary, FIA data and local forest reconstruction studies suggest that basal areas are broadly similar to, or somewhat higher on average than basal areas in presettlement times. Local trends will depend on the relative effects of fire suppression (which would increase basal area on its own) versus logging and other forest mortality agents (which decrease basal area). It is important to remember that most (but not all) reconstruction studies are done in areas that did not suffer logging, whereas the FIA dataset samples the forest without bias.

Future:

Miller and Urban (1999b) simulated the effects of climate change on forest biomass and composition along an elevational gradient in Sequoia-Kings Canyon National Parks (see Forest Composition and Species Diversity for more detail). At their two lowest elevation sites, which fall in YPMC forests, woody biomass was completely or almost completely lost by the end of the simulations. Under their less extreme warming and drying scenarios, basal area at 1800 m declined from 29 m²/ha to 4 m²/ha, and was completely lost under the more extreme scenarios. At 2200 m, basal area declined from 52 m²/ha to 25-32 m²/ha under the less extreme future scenarios, and to 8 m²/ha under the most extreme scenario. Miller and Urban (1999b) included fire in their simulations, but only surface fires were modeled, so fire played little role in directly causing biomass loss. Miller and Urban's (1999b) results are thus probably conservative with respect to the velocity of change in biomass.

Bachelet et al. (2001) modeled the potential effects of climate warming on carbon budget using climate data from seven different Global Circulation Models. Their results varied widely, depending on the GCM used, the vegetation/carbon model used, the degree of future warming, and precipitation. Forested areas were especially sensitive to changes in temperature. Bachelet et al. (2001) found that most scenarios predicted increases in forest cover and forest biomass in the assessment area under moderate warming (+2-4° C); their vegetation model that included fire resulted in less biomass gain. Their projections included a relatively large increase in precipitation, however according to Dettinger (2005), the most common prediction among more recent GCMs (which are considerably more complex and realistic than the models available in the late 1990s) is temperature warming by about 5 °C by 2100, with precipitation remaining similar or slightly reduced compared to today. The newer GCMs thus cast some doubt on Bachelet et al.'s (2001) results. Under more extreme warming scenarios (≥+4° C), Bachelet et al. (2001) found that increases in forest biomass were reversed. They referred to this temperature-driven pattern as the "early green-up, later browning" hypothesis. Overall, Bachelet et al. (2001) suggested that a temperature increase above a threshold of about 4.5° C would result in more carbon loss than sequestration across the US. In a subsequent modeling effort, Bachelet et al. (2007) projected that biomass consumption by wildfire would increase by 25-67% across most of the assessment area over current (1961-1990) rates.

Lenihan et al. (2008) modeled vegetation distribution and productivity in California under three future climate change scenarios. Under the most moderate warming scenario, which also featured similar precipitation to today, the extent of tree-dominated vegetation in the assessment area increased substantially (although some losses of conifer forest occurred) and California as a whole was modeled to be a significant carbon sink by the end of the 21st century. Under the two drier and warmer scenarios, significant loss of conifer forest occurred (much of it to fire), and broadleaf expansion was not sufficient to offset a large net loss of ecosystem carbon by 2099.

Canopy cover

Canopy cover is a measured of the percent of the ground surface that is shaded from directly above by trees. A tree's "canopy" is a polygon whose perimeter coincides with the outer edge of the tree's foliage. The canopy is assumed to be whole, and gaps within the tree canopy perimeter are not subtracted. In this section we refer to "relative" canopy cover, where overlapping

tree canopies are not double counted (or triple counted, etc.), so the maximum canopy cover in a plot is 100%. Canopy cover is usually an overestimate of the amount of shade on the underlying ground, because it ignores intercanopy gaps. “Canopy closure” is better correlated with actual light availability, but it must be measured on a plot by plot basis and cannot be easily modeled or estimated from plot tree data (see North and Stine 2012).

NRV:

As indicated in the forest structure section, observations of forest cover in the 19th and early 20th century are overwhelmingly characterized by remarks as to the general openness of the canopy of YPMC forests in and near the assessment area, but it is clear from descriptions that conditions varied across the landscape (e.g., King 1871, Muir 1894, Sudworth 1900, Leiberg 1902, Greeley 1907, Larsen and Woodbury 1917, Brewer 1930; also see Pearson [1923] for descriptions of yellow pine forests in Arizona). Recently a number of researchers have undertaken plot-based reconstructions of stand structural conditions in the late 19th century, and data are also now available from a number of contemporary reference sites, where logging did not occur and – in many cases – where fire was not completely suppressed during the 20th century. From these studies we can now obtain a more quantitative idea of the levels of canopy cover supported by YPMC forest stands before timber harvest and fire suppression altered much of the landscape.

Historical datasets and reconstructions of presettlement or pre-fire suppression conditions do not generally report canopy cover. However, most of these datasets provide data on size class distributions of trees in the sampled plots. Where such data were available, we used two methods to generate canopy cover estimates. First, we used species-specific equations for crown-width (Table 4.4.2.1 in Keyser 2010) from the Forest Vegetation Simulator (Dixon 2002) to estimate crown area, using the midpoint of the size class in our calculations, then multiplying by the number of trees in that size class. Percent cover was obtained by summing across the size classes and standardizing to square meters, then dividing the result by 10000 (number of m² in a hectare) and multiplying by 100 for percent. Second, cover was adjusted by applying a formula that assumes random spatial location of the trees in the sampled area and subtracts overlapping canopies from the maximum cover value (Crookston and Stage 1999). In Table 4 we report both the adjusted value and the value before adjustment, which assumes no trees in the stand grow beneath any others and may therefore be seen as a rough estimate of the maximum possible canopy cover in the sampled stands.

Direct measurements of tree canopy cover in contemporary reference yellow pine and mixed conifer forests, and modeled canopy cover from historical datasets both strongly substantiate the oft stated notion that presettlement forests were generally more open than modern forests in the assessment area (Table 4). Average canopy cover values ranged from around 17% to less than 50%, except in the Sudworth (1900) dataset (16 YPMC plots from his Table 1), which averaged almost 70%. The much higher canopy covers in Sudworth’s data support the notion that his data were not a random sample of forest conditions (Stephens and Elliott-Fisk 1998, Bouldin 1999, Stephens 2000). Uncorrected modeled cover values suggested that values as high as 70-90% were possible in very dense forest stands (Table 4).

With respect to the FVS modeled values, it should be noted that a few studies have found that FVS techniques tend to estimate lower cover values than field-based methods, especially in areas of higher canopy cover. Fiala et al. (2006), in Douglas-fir/western hemlock forests in western Oregon, found that FVS estimated lower canopy cover than four field-based methods by 11-27% in high canopy cover, mature and old growth forests. The sampled stands were almost entirely >60% cover however, and the difference between FVS and the most precise field methods became nearly zero below 60% canopy cover. Since YPMC forests in the assessment area were highly heterogeneous and mostly open canopied, FVS modeling probably provides a reasonable “ballpark” estimate of canopy cover for historical and reconstruction plots. Another issue is that most of the modeled FVS values in Table 4 were carried out on summary data that had been organized into size classes. This lessens the variability of the input data and could affect the calculated mean in either direction, depending on the distribution of tree sizes within the size class categories.

Comparison to current:

Data from the compiled FIA plots (USFS 2013) show that mean modern canopy cover in YPMC forests is 45.7% (+/- 21.8 st. dev.; westside mean = 50%, eastside = 38%). The average from the 13 reference values in Table is 36.8% (+/- 13.6 st. dev.) when including Sudworth, 34% when excluding Sudworth. The FIA canopy cover values were generated using the same FVS algorithm that produced most of the values in Table 4. The modern average is about 25% higher than the presettlement average from Table 4, but, as with all of the quantitative comparisons using the FIA data, it is difficult to statistically compare the two data sources given their different scopes and sample sizes. Generally however, the data lend strong support to the idea that current canopy cover is higher on average than under presettlement conditions.

Future:

Assuming current trends in forest densification and fire suppression continue, canopy cover will probably continue to increase across the YPMC forest belt. This will be offset to some extent if forest mortality agents, such as fire, insects, and disease, increase.

Forest gaps and tree clumps

NRV:

A “forest gap” is an opening or hole that exists within the forest canopy, or put another way, an area within a forest stand that is unoccupied by mature trees. Forman (1995) argued that gaps are small entities that exist within a mosaic of larger landscape “patches” (or what foresters refer to as “stands”), where a patch is a relatively large and homogeneous area that differs from its surroundings in its structure and composition. Essentially, patches are a component of coarse-grained heterogeneity, whereas gaps are a component of fine-grained heterogeneity. In assessment area YPMC forests, the historical rarity of large-scale severe disturbances resulted in a very fine-grained structurally heterogeneity that does not lend itself well to the recognition of landscape patches (Franklin and Fites-Kaufman 1996, USDA 2001). Given the generally low canopy cover and clumpy nature of YPMC forest stands, many presettlement forest stands were as dominated by gaps as by trees.

When gaps are created in the canopy, enhanced light is provided to the forest understory, which often results in a dynamic ecological response from both plants and animals. Gaps are important locations for tree, shrub, and herb regeneration, and they also serve a variety of roles in providing cover, forage, and other habitat for animals. Gaps are created through disturbance processes (“disturbance patches”), or they can also arise as a consequence of the physical habitat, e.g. soil type, rocks, and so on (“environmental patches”). The sizes and distributions of canopy gaps in forested ecosystems are important ecological variables (Connell 1989, Spies and Franklin 1989, Agee 1993), and over the last few decades it has become clear that the creation and maintenance of canopy gaps is an important forest management objective. Since open gaps in the forest canopy in a reasonably productive forest will quickly fill with regeneration, the size of “regeneration patches” can be roughly equated to the sizes of the original disturbance patches that led to the regeneration event. After growth to adult size, the surviving members of the regeneration patch will form an aggregation of trees, or a “tree clump” (Agee 1993). In general, canopy gaps are somewhat larger than the regeneration patches that arise within them, and mature tree clumps somewhat smaller than the regeneration patches that gave rise to them (Stephenson 1999).

Larson and Churchill (2012) outlined three components of local pattern in frequent fire yellow pine and mixed conifer forests: tree clumps, widely spaced individual trees (usually large), and openings. Tree clumps could be comprised of similar aged overstory trees, multi-aged aggregates, or they could be regeneration patches. In the studies they reviewed, the average number of trees in a clump ranged from 2-44 trees, and the average number of clumps per hectare ranged from 10-27 (Larson and Churchill 2012).

A number of studies in and around the North American Mediterranean-climate zone have used stand mapping techniques to estimate typical sizes of canopy gaps, regeneration patches and tree clumps in YPMC forests. Table 5 provides a summary of the range of published gap-patch-clump sizes in the literature. Minimum sizes averaged about 0.04 ha, maximum sizes averaged about 0.3 ha, with a maximum range from 0.07 to 1.17 ha. Larson and Churchill (2012) carried out an independent summary of tree clump sizes in yellow pine and mixed conifer forests across the western US, and found a range of 0.003 to 0.4 ha per clump. Their summarized range for regeneration patches was somewhat wider, but with a similar mean: 0.001-0.64 ha.

Future:

Increased forest densification will continue the trend of reducing forest openings/gaps. This will be offset to some extent if forest mortality agents continue to increase.

Snags and coarse woody debris (CWD)

NRV:

As he left the Sierra Nevada after three years of travel and mining, Peter Decker, in his diary from 1849-1851, wrote that dead trees were seldom seen and “it is very rarely that timber is laying rotting” (Giffen 1966, p 232). Arno (2000, p 100) stated that, in western YPMC forests characterized by understory fires, “shrubs, understory trees, and downed logs were sparse, as

testified to by dozens of historical photographs and narrative accounts.” In areas characterized by more moderate/mixed severity fires, surface fuels and CWD tended to be heterogeneously distributed, with concentrations found in scattered patches (Arno 2000). Skinner (2002) wrote that, because YPMC forests in California burned so often, “It is unlikely that much large woody material survived fire long enough to decompose fully in fire regimes that preceded the fire-suppression era.”

Stephens (2004) reported on fuel loads and snag densities in unlogged YPMC forests in the Sierra San Pedro Mártir (SSPM), which have also only seen recent suppression of fires. Stephens (2004) found that snag density in the sampled forests (all stems ≥ 2.5 cm dbh) averaged 3.5 per ha before a major drought, and 5.1 per ha after a multiyear drought. Heterogeneity in snag distribution was very high: the modal snag density in the first sample was zero per ha (35% of plots), in the second sample it was about 2.5 per ha (27% of plots had no snags). The average snag measured 59 cm (23.2”) dbh. CWD, measured as the 1000 hr fuels (branches and logs > 7.6 cm [3”] dbh), averaged 15.8 tons/ha, but more than a third of plots had no CWD and only 20% of the plots supported 75% of the CWD. 81% of the CWD was in a rotten state, 19% sound (Stephens 2004).

Stephens et al. (2007) reported on CWD over 15 cm (5.9”) in diameter and ≥ 1 m long from the SSPM. Thus defined, CWD averaged 15.7 tons/ha, with a large range (0-154.5 ton/ha). The median was only 1.4 tons/ha, and about half of the plots sampled contained no CWD of this size or above. Overall, the average density of CWD was about 108 pieces/ha (+/- 16.3 SE). Most sampled logs were rotten.

Lydersen and North (2012) sampled YPMC forests in the assessment area that had not been logged and had experienced at least two fires in the 65 years before the field work commenced. They found very high variability in snag and CWD measures. Snag density for snags greater than 50 cm (20”) dbh averaged 9.1/ha, and snag volume averaged 123 m³/ha (+/- 20.1 SE), with the highest densities and volumes of snags occurring on lower, northeast facing slopes. Lydersen and North (2012) provide two measures of CWD: logs ≥ 50 cm in diameter averaged about 10.3 pieces/ha (no measure of variability given), while 1000 hr fuels (> 7.6 cm diameter) averaged 32.3 tons/ha (+/- 3.4 SE).

Agee (2002), in a paper discussing the general relationship between fire regime and the creation and persistence of snags and coarse woody debris, suggested that a Fire Regime I forest might typically support around 5 snags/ha, with the average snag size about 75 cm (30”) dbh. According to Agee (2002) CWD derived from the same average size of tree might total around 5 tons/ha. Because of very frequent fire in these forest types, both snags and CWD would fall and/or decompose at a higher rate than under conditions free of fire. For mixed severity fire regimes, Agee (2002) estimated that snag densities and CWD mass would be higher, as fires were more severe, which creates more snags and CWD, and less frequent, which allows them to persist longer on the landscape. Agee cited Wright (1998), who found an average of 40 tons/ha of snags and 55 tons/ha of CWD (ranging from 20-250 tons/ha) in Douglas-fir forests. Agee (2002) noted that modern fire-suppressed forests that used to experience frequent, low severity fire might now support (much) more CWD than they did under presettlement conditions. Concerning these Fire Regime I forests, Agee (2002), stated that:

“Burning prescriptions designed to retain most coarse woody debris can produce a “non-window”: duff moisture levels so high that such moisture contents are rarely if ever attained on these sites. Constraints to preserve all or most coarse woody debris effectively eliminate the use of fire for restoration purposes and leave the dry forest types at risk for stand-replacing fire. When such high-severity fire occurs, it brings with it the “boom and bust” coarse woody debris dynamics of the high severity fire regimes. This is a classic case of the fine-filter (log preservation) trumping the coarse-filter (restoring the natural process), and in the long run is likely to result in a failed conservation strategy.”

Harrod et al. (1998) used stand reconstruction and modeling techniques in an attempt to derive historic reference conditions for snag densities for an area of dry ponderosa pine/Douglas-fir forest in eastern Washington. They determined that, depending on the size classes included, snag densities under presettlement conditions probably ranged from about 14.5 to 34.6 snags/ha. Agee (2002) noted that Harrod et al. (1998) did not incorporate the effects of fire on snagfall rates, implying that their numbers were probably biased upward.

Youngblood et al. (2004) studied three areas of old growth ponderosa pine forest in eastern Oregon and northern California. All sites had experienced about a century of fire suppression, although parts of the California site had recently experienced a prescribed fire. Youngblood et al. (2004) found that snag densities ranged from 8 to 12 per ha (trees ≥ 15 cm dbh), and CWD (≥ 15 cm diameter and 1 m long) averaged about 47 pieces/ha (± 5.3 SE). 85% of the CWD was between 15 and 65 cm in diameter, and most pieces were rotten (i.e., old). Youngblood et al. (2004) noted that the long-term demography of both snags and CWD in western frequent fire forests is largely unknown, with the current lack of fire in these stands a particular impediment. They suggested that, with respect to their ability to discern presettlement reference conditions from their data, fire exclusion over the previous century had probably biased their measurements of CWD (and snags) upward.

Comparison to current:

Stephens (2004) provides a table (Table 4, p. 109) summarizing snag densities for seven studies in areas managed under fire suppression during the 20th century, two of which had also experienced timber harvest. The minimum sizes of measured snags vary from study to study, but the average snag density from these seven sites is 47.9 snags/ha, an order of magnitude higher than found in Stephens' (2004) study of the Sierra San Pedro Mártir, and also much higher than two other studies of unlogged YPMC forests also summarized in the table (Savage 1997, Stephens 2000).

Studies of the effects of prescribed fire on CWD and snags in the assessment area provide evidence that levels of both were lower in presettlement forests than in current, unlogged forests. Kauffman and Martin (1989) summarized the results of 60 prescribed fires carried out at three experimental sites in the northern Sierra Nevada in the mid-1980s. Fires were ignited at four different times of year: early and late spring, and early and late fall. We summarize the results from late spring and early fall burns, as they best represent the natural fire season in the assessment area. Fires reduced the mass of sound CWD (>7.6 cm diameter, i.e., 1000 hour fuels) by

62% on average, and the mass of rotten CWD by 87%, resulting in an overall postfire mean of 11.3 tons/ha; CWD averaged about 28 tons/ha before fire.

Various other studies of the effects of prescribed fires on CWD and snags corroborate the Kauffman and Martin (1989) results. For example, Kilgore and Sando (1975) measured 15 tons/ha of 1000-hr CWD after fire (down from 97.5 ton/ha prefire); Stephens and Finney (2002) measured 3.4 tons/ha (from 42.1 tons/ha prefire); and Knapp et al. (2005) reported 15 tons/ha postfire (from 66.2 tons/ha prefire) for their fall burn. Kilgore (1973a) measured the change in log (>15 cm diameter) mass after prescribed fire, finding a reduction to 2.8 tons/ha from 12.8 tons/ha prefire. Knapp et al. (2005) also measured losses in log mass (>15 cm diameter) to a fall prescribed fire: postfire = 7.4 tons/ha, prefire = 47.2 tons/ha. Kilgore (1973a) reported that snags were almost entirely reduced to charred stumps and cavities in the prescribed fire he studied. Uzoh and Skinner (2009) investigated the effects of prescribed fire on CWD (≥ 7.6 cm diameter) at Blacks Mountain Experimental Forest in the northern subregion of the assessment area. A single fall burn reduced CWD mass by 1.5 to 2 times over the mechanical treatment effects, which also reduced CWD. In the burned plots, overall reductions of CWD ranged from 57% to 90%, with higher losses in the more decayed CWD classes. Uzoh and Skinner (2009) concluded that “restoration goals that include large amounts of decayed, coarse woody debris do not appear to be appropriate for these dry pine forests of northeastern California.”

An important point to be made is that these large reductions in CWD (and snags) were the result of single prescribed fires under moderate conditions at times of the year when naturally ignited fires are uncommon because of higher fuel moistures. They were also all carried out in stands that had not experienced fire for 50-100 years or more. Average fire return intervals in presettlement assessment area YPMC forests were between 10 and 20 years.

The most recent FIA data compilation (USFS 2013) gives an average of 23 tons/ha (+/- 12.9 st. dev.) for CWD measured as 1000-hr fuels (≥ 7.6 cm diameter) in contemporary YPMC forests. The average of Stephens (2004), Lydersen and North (2012), Kauffman and Martin (1989), Kilgore and Sando (1975), Stephens and Finney (2002), and Knapp et al. (2005), all of which we interpret to broadly represent reference conditions, is 15.5 tons/ha (+/- 9.4 st. dev.). The FIA cutoff for measurement of logs is a diameter of 25 cm, which does not perfectly correspond to any of the studies we found in the scientific literature. Nonetheless, the FIA average density for logs ≥ 25 cm in diameter is 42.6 pieces/ha, which is comparable to the density of 44.8 pieces/ha (logs ≥ 30 cm diameter) found by Stephens et al. (2007) in the Sierra San Pedro Mártir.

FIA data give an average snag density (≥ 15 cm dbh) of 37.2 snags/ha (+/- 69.5 st. dev.; mixed conifer mean = 47.7, yellow pine mean = 20.2) for contemporary YPMC forests in the assessment area. For the same size categories, Stephens (2004) found 4.4 snags/ha in the reference forests in the Sierra de San Pedro Mártir, and Youngblood et al. (2009) found 8-12 snags/ha in the old growth pine forests they studied. Harrod et al.'s (1998) estimate of presettlement yellow pine forests in eastern Washington, which did not account for fire consumption of dead trees (see above), was in the range 14-36 snags/ha.

Based on these data sources and considerations, we conclude that the levels of CWD and snags in modern YPMC forests in the assessment area are somewhat higher on average than in the average presettlement YPMC forest stand.

Future:

Continuation of current trends toward more snags and CWD seems likely, especially if non-fire forest mortality agents increase substantially. Increases in fire frequency will increase snags and CWD initially, but will reduce them as areas are reburned.

Forest understory and nonforest vegetation

Few of the early observers of YPMC forest in the assessment area provide more than passing reference to nonforest vegetation, except with respect to seedling and sapling densities and an occasional reference to shrubs.

Tree seedlings and saplings

NRV and comparison to current:

The very low average density of tree seedlings and saplings in YPMC forests alarmed early foresters in the assessment area, but it was clear to them that recruitment potential was high and fire was the main factor reducing recruitment (e.g., Sudworth 1900, Leiberg 1902, Greeley 1907). Sudworth (1900) wrote that:

“The frequent open spaces in yellow-pine forests are sooner or later covered with dense patches of young trees, but these thickets may in turn be swept off by fire. So continuous and widespread are these forest fires that... they keep a very large percentage of the seedling growth down... The forest floor looks clean swept. But the remarkable productive power... is seen only in localities where fences and the exclusion of fire have protected the incoming seedlings. Here the stand is so dense as to be quite impenetrable.”

Where seedlings and saplings successfully survived (usually due to longer intervals without fire), shade tolerant species tended to dominate. Greeley (1907) wrote:

“The very large proportion of fire and cedar in its reproduction is the worst feature of the west Sierra Forest. Both of these species are prolific seed bearers... Except at lower elevations and on very warm exposures, where yellow pine grows in pure stand, dense thickets of fir and cedar crowd the young pine down to one fifth or less of the reproduction.”

Bonnicksen and Stone (1982), in one of the first modern stand reconstruction studies, noted that the 1890s landscape at their study site supported much more open ground with seedlings and saplings than the current forest. Around 50% of the area covered by seedling and sapling “aggregations” was dominated by white fir, which surprised the authors, as they had expected shade intolerant species to dominate recruitment into gaps.

What emerges is an ecosystem where conditions were greatly determined by the frequency and nature of fire. Frequent fire substantially reduced reproduction and the resulting bare mineral soil and open forest conditions favored species like yellow pine and black oak, which could bet-

ter survive low intensity burns and did better under high light conditions (Pearson 1942, Burns and Honkala 1990, Moghaddas et al. 2008, Barbour et al. 2007).

Modern reference sites like the Sierra de San Pedro Mártir or Beaver Creek Pinery (Lassen National Forest) show very high heterogeneity in seedling densities, which is similar to the conditions described by early observers. The mean seedling density in the Sierra de San Pedro Mártir was measured by Stephens and Gill (2005) at 125 seedlings/ha, with about 50% of the sampled plots supporting <100 seedlings/ha and 10% supporting >300 seedlings/ha. Mean seedling density in the Beaver Creek Pinery was measured by Taylor (2010) at about 660 seedlings/ha, including ponderosa pine and black oak. Over 2/3 of the ponderosa pine seedlings were found in high light conditions in forest gaps and variation in densities was very large, ranging from 37 to 1408 seedlings/ha.

Compilation of the most recent FIA data (USFS 2013) gives a mean of 1820.6 seedling/ha (+/- 3973 st. dev.) for yellow pine and mixed conifer plots. The coefficient of variation (CV) is 2.2, which indicates high variance in seedling density among plots. We conclude from the data and from inference based on successional processes that current mean seedling densities are almost certainly higher than mean seedling densities under presettlement conditions.

Future:

Seedling composition will largely reflect the overstory composition (van Mantgem et al. 2006). The very high proportion of shade tolerant species in current YPMC forests in the assessment area suggests that future forest recruitment will be dominated by species like white fir and incense cedar. Even where repeated prescribed fires have been employed in order to enhance pine and oak recruitment, the persistence of mature shade tolerant species in the forest canopy has resulted in a strong dominance of these species in postfire seedling densities (Webster and Halpern 2010). If an increase in recruitment of fire and drought tolerant species is desired, wholesale removal of shade tolerant species from the overstory and/or direct planting of fire tolerant seedlings may be necessary.

Shrubs

NRV and comparison to current:

Like tree cover, most evidence is that shrub cover in presettlement YPMC forests was relatively low but highly heterogeneous on the landscape. For example, Fitch (1900) described forests in the Yosemite area as: "...remarkably free from undergrowth... and only along streams, in the bottom of gulches, and on rocky southern slopes is the brush so thick as to impede progress." Marshall (1900), surveying the higher elevations of Yosemite for the USGS, which included some areas of moist mixed conifer, stated that the forest was "everywhere open without undergrowth of any kind." Decker, in his journal, wrote of the forests in the upper Mokelumne River drainage that there was "generally no underbrush" (Giffen 1966). On the other hand, Bruff, in his journal from the same time period refers to both open forests and forests with dense, bushy undergrowth (Read and Gaines 1949).

Leiberg (1902) stated that much of the forest understory in his survey area was more or less bare. At the same time, he noted that burned areas supported a “great amount” of undergrowth, mostly dominated by species of *Ceanothus*, which he ascribed to recent anthropogenic fire activity. According to Leiberg, burned areas in YPMC forest often supported dense growths of montane chaparral, from four to five feet (1 to 1.5 m) in height. He was convinced that montane chaparral came only from fire in previously forested stands and counted such areas as representing destroyed forest even where there wasn’t any evidence of previous forest. Greeley (1907), referring primarily to fires set by Euroamericans, decried the loss of valuable timberlands to montane chaparral.

Mitchell (1913) noted that the quantity of brush (shrubs) in yellow pine forest tended to minimal, but it was much higher in mixed conifer stands (and even higher in red fir forests). Given that forest openings were much more prevalent in yellow pine stands, the difference was probably due to more frequent fire (which reduced shrub cover), less intense fire (which would result in less fire-induced germination of shrubs like *Ceanothus* or manzanita), and lower site productivities.

Show and Kotok (1924) decried the loss of forest to severe fires set by man, especially in situations where logging slash was heavy. Severe forest fires in the assessment area normally result in postfire succession to shrublands (usually some form of chaparral), especially on the west side. In the absence of further fire, such shrublands will last for decades before succession to forest occurs. In the presence of further fire, such stands can become quasi-permanent features of the landscape, since chaparral generally burns at high intensity and kills most young trees (Nagel and Taylor 2005, Skinner and Taylor 2006). Show and Kotok (1924) produced a table providing summaries of the areas of the ten northern National Forests (minus the Modoc) that supported “brushfields” in the early 1920s. In their interpretation, these brushfields had developed in previously forested stands after fires set by Euroamerican settlers. We reproduce that table for the six National Forests in the assessment area in Table 6, and compare it to modern areas of chaparral on the same National Forests. Percentages of National Forest area in brushfields in the early 1920s ranged from 7% on the Eldorado National Forest, to over 16% on the Stanislaus National Forest; the overall average was about 11%. Modern data (from the most recent Forest Service existing vegetation mapping, data at <http://www.fs.usda.gov/detail/r5/landmanagement/gis/?cid=STELPRDB5327836>), using “mixed chaparral” and “montane chaparral” CWHR types where they occur in sites identified as having sufficient productivity to support forest as our definition of shrublands, show that the overall averages are slightly lower on average. Four of the National Forests experienced a decrease in the area occupied by shrubs, and two experienced increases. According to our comparison, the Eldorado National Forest experienced the greatest increase (+31%), the Plumas National Forest the greatest decrease (-47%). It is important to note that the most recent vegetation mapping on the northern Sierra Nevada National Forests (Lassen, Plumas, and Tahoe) was completed some time ago, using imagery from 2000 and 2005 (C. Ramirez, pers. comm.). The values in Table 6 thus exclude the large expanses of fire-caused shrubfields that have resulted from the high number of large, often very severe fires that have occurred in the northern Sierra Nevada since 2005 (e.g., Rich, BTU Complex, Moonlight, Antelope Complex, Reading, American River Complex, Ralston, etc.). Including these numbers would raise the modern percentages substantially, especially on the Plumas National Forest. It is also important to note that the techniques for measuring the areal extent of shrub-

fields in the two surveys were different (summary of ocular and land surveyed measurements in 1920s, vs. remotely sensed imagery in 2000s), and the numbers in Table 6 should therefore be viewed as approximations.

Modern studies of YPMC forests have found an average of about 15-25% relative shrub cover (but with high variability) in reference forests that have not been logged and/or have not experienced extreme fire suppression (Table 7). Note that very large areas of chaparral would not have been sampled by these studies as they would not be considered forest for sampling purposes.

Contemporary FIA data on shrub cover are difficult to compare with other datasets, as shrub cover is computed by summing the species-specific covers without accounting for overlap (and therefore total cover can sum to more than 100%). Average absolute cover by shrubs in YPMC forest plots is 22.6% (+/- 23.3 st. dev.; USFS 2013). These values will be a bit higher than the actual relative values.

Duncan Dunning established three 4 ha (10 ac) forest plots in the Stanislaus-Tuolumne Experimental Forest in the 1920s that have been recently resampled by the PSW Research Station. Peer-reviewed publications are forthcoming, but Parks (2009) notes that shrub cover in the original plot data is about 30%, but currently shrubs cover only 3% of the sampled forest stands. Bonnicksen and Stone (1982) carried out a stand reconstruction in a small watershed of YPMC and giant sequoia forest in Kings Canyon National Park. They estimated that about ¾ of the forest aggregations dominated by large and very large trees had some sort of shrub cover in their understory in 1890. They also found that about 19% of the watershed was covered by shrub dominated “aggregations” in 1890, vs. 11% in 1977, but these would represent shrubfields rather than forest understory.

Vankat and Major (1978) found that, in general, shrub cover had dropped in YPMC forests in Sequoia-Kings Canyon National Parks over the previous half century or more. They ascribed the changes to more light competition from trees, increased browse by ungulate populations, and/or less fire. The most affected species were manzanita and *Ceanothus*, both of which support species with fire-cued germination. Vankat and Major (1978) also noted some areas where shrub cover had increased.

Laudenslayer and Darr (1990), studying areas where timber harvest had occurred, stated that shrub cover had increased in most places due to greater availability of light and the long-term lack of fire, which they suggested had reduced shrub mortality.

Comparison of aerial photos from 1930s and 1940s with photos from today sometimes shows greater cover of shrubland in the early photos. It is difficult to determine whether the early photos are NRV however, as there were many severe fires set by settlers in the late 1800s. For example, Nagel and Taylor (2005) found that the average area of six chaparral stands in the Lake Tahoe Basin retracted by 62.4% between 1939 and 2000. It is important to note that these comparisons are mostly focused on large stands of chaparral, rather than patches of shrubs found within the forest matrix itself.

Thorne et al. (2008) compared Forest Service vegetation maps from the 1930s (VTM project; Wieslander 1935) with modern Forest Service vegetation maps in the lower elevations of El-dorado County, on the west side of the central assessment area. They found that the extent of

montane chaparral stands had declined by over 90% over the sixty year period. Thorne et al. (2008) noted that some of these stands had potentially transitioned to hardwood stands, but others were large patches of chaparral from earlier timber harvest and fires that were reoccupied by conifer forest after the institution of fire suppression. As above, the extent of chaparral on the landscape in the 1930s was probably notably enhanced over presettlement conditions by human activities.

Kauffman and Martin (1990) reported on the effects on shrubs of 60 prescribed fires in the northern Sierra Nevada. We report only those results from the late spring and early fall burns (N = 30), as they best replicate the seasonality of natural fires. Kauffman and Martin (1990) found that fires at these times of year tended to consume more fuel and (mostly) burned hotter than fires in the early spring and late fall. One year after fire, an average (across the three study sites) of 25% of shrubs within the burn plots had survived the early fall burns, and 36% survived the late spring burns. Kauffman and Martin (1990) also tracked two year survival of black oak and tan oak (*Lithocarpus densiflorus*) and found only 9 and 18% survival (black oak and tan oak, respectively) in the early fall burn and 28 and 21% survival from the late spring burn. Survival was higher from the early spring and late fall burns. Seedling recruitment and sprouting from *Ceanothus* species was relatively strong, with hotter burns increasing mortality and reducing sprouting but increasing the number of seedlings. Kauffman and Martin (1990) concluded that hot burns could be used to reduce shrub cover on sites where that was a management goal.

Overall, considerations of shrub cover on assessment area landscapes suggest that the overall portion of the YPMC landscape occupied by shrubs today is broadly similar to, but possibly somewhat lower than the portion occupied at the beginning of the 20th century. Many early observers believed that shrub cover had been increased after Euroamerican settlement, so it may be that the current overall areal extent of shrublands is actually somewhat higher than when Euroamericans began arriving in the mid-19th century, but we have no quantitative data to substantiate this inference. At the same time, fire suppression, logging, and other management practices since the beginning of the 20th century have greatly homogenized YPMC forests in the assessment area, resulting in higher canopy cover and denser stands, and less optimal conditions for shrub survival in the forest understory. In summary, the landscape extent of shrubfields and other early seral vegetation within the YPMC forest belt is probably within NRV, but shrub cover in the forest understory has probably been generally reduced by increasing tree density and canopy cover and decreasing understory light availability.

Future:

The increase in fire area, fire severity, and high severity patch size in assessment area YPMC forests over the last quarter-century (Miller et al. 2009, Miller and Safford 2012, Mallek, et al., in prep.) has led to a recent increase in the extent of early seral montane chaparral stands, especially in Forest Service lands in the northern half of the assessment area. Most future models and analyses of paleo-data suggest that future disturbance cycles and warming climates will further increase the amount of early seral vegetation on assessment area landscapes (e.g., Lenihan et al. 2003, 2008; McKenzie et al. 2004, Cole 2010).

Grass and forbs

NRV and comparison to current:

William Brewer's (1930) memoirs of his 1861-1864 travels with the Whitney Survey refer to the scarcity of good grass cover in the Sierra Nevada, and lush areas of grass are highlighted where they occur. Sudworth (1900) also referred repeatedly to the lack of good pasturage, and states toward the beginning of his report that "...forage is exceedingly short on all the unfenced mountain ranges." Later Sudworth suggests that in many places the lack of herbaceous growth is due to the effects of sheep grazing, although it is important to note that his field surveys also coincided with one of the worst California droughts on record. Interestingly, Fitch (1900), working adjacent to Sudworth's survey area, described "excellent pasturage" on high mountain slopes, meadows, and river valleys, as well as in open timber stands. None of Sudworth's photos from the YPMC forests, except perhaps one from the lower boundary with the oak-digger pine belt, which supports much annual grassland, show any obvious areas of grass cover. Leiberg (1902) noted that humus layers were rare to nonexistent in the forests he surveyed, the forest floor was usually bare, perhaps with a layer of needles "rarely exceeding two inches in depth".

Evetts et al. (2006, 2007), studied soil phytoliths in mixed conifer forests in the North American Mediterranean zone, looking to determine whether there was any support for hypotheses of substantial grass cover in presettlement YPMC forests. Phytoliths (also known as "grass opal") are small structures of silica left behind in the soil after death of the parent plant, and grass phytoliths are morphologically unique. Evetts et al. (2006) found that the grass phytolith content of soils in a giant sequoia-mixed conifer forest in the southern Sierra Nevada indicated a long-term (at least centuries) lack of a substantial grass component at the study site. The study found evidence of somewhat higher grass cover in some places on the landscape however, such as in ponderosa pine-dominated forest on ridge tops, and in areas near stream channels. Evetts et al. (2007) carried out a similar study in a Jeffrey pine-mixed conifer forest in the Sierra San Pedro Mártir of northern Baja California. As above, they found that grass phytolith levels were too low to indicate extensive cover of grass over the previous centuries.

Takahashi et al. (1994) studied volcanic soil development under mixed conifer forests in northern California. Presence of melanic epipedons (humus) in forest andisols has traditionally been ascribed to periods in which the soils supported grassy vegetation, but phytolith analysis of the California YPMC soils showed very little to no grass component could have been present during the formation of the soils, i.e., over many centuries. According to Takahashi et al. (1994), the occurrence of frequent fire in their study forests may have been the key factor in forming melanic epipedons in the andisols they studied.

Modern surveys of YPMC forests in the California Mediterranean zone do not suggest that grass or forb cover is particularly high, even in relatively undisturbed stands with low canopy cover. Smith (1994) described 45 associations of yellow pine forests in the northeastern assessment area, sampling only from uncut stands that were at least 100 years old. Few of the sampled sites were heavily grazed (S. Smith, pers. comm.). Average overstory cover (relative) was about 53% (range = 13-82%), shrub cover 21.4% (range = 1-61%), forb cover 8.9% (range = 2-41%), and grass cover 6.5% (range = 2-20%). Fites (1993) carried out a similar survey of older and undis-

turbed mixed conifer forests in the northern and southern Cascades. Canopy covers were much higher in these more productive sites (range = 25-96%), and forb and grass covers similarly low (relative cover means of 11.8% and 3.2% respectively). Oliver (2000) described forest conditions at Black Mountain Experimental Forest on the Lassen National Forest. Perennial grass and herb cover was very low both before and after forest treatment (<4% total in both cases), although the cover of exotic invasive annual grasses increased notably after treatment, especially prescribed fire. Barbour et al. (2002) sampled uncut old growth mixed conifer stands in the Lake Tahoe Basin and found an average of only 0.5% (range = 0-10%) relative herbaceous cover, even though overstory tree cover was only 45% on average.

The contemporary FIA data provide measures of herbaceous cover that are generated by summing the cover percentages of the different herbaceous species, thus total cover can sum to more than 100%. Comparison with relative cover data (which is what is typically reported in scientific studies) should be done with care, as the FIA summing practice ignores overlap among plants. The average cover of herbaceous plants reported from the FIA plots is 11.1% (+/- 12.5 st. dev.) (USFS 2013). It is important to remember that the FIA dataset is a statistical sample of all stand conditions, whereas the reference values measured above are all from relatively undisturbed, old growth-type stands.

The Sierra de San Pedro Mártir (SSPM) in northern Baja California has not been logged and has experienced more or less effective fire suppression since the 1970s or 80s. Cattle are present but spend little time in the upland forest at any distance from water sources and meadows. A recent study of the understory vegetation in conifer stands in the SSPM found that relative forb cover averages 9% and grass cover less than 2%, under an average tree cover of 29%. Drier and lower elevation Jeffrey pine stands in the Sierra Juárez near the US border show similarly low levels of understory cover: averages of 6.6% and 11.8% for forbs and grasses, respectively, under tree cover of 36% on average (Safford, unpub. data).

Pearson (1942) described the effects of herbaceous vegetation on ponderosa pine recruitment and survival, and explained how the substantial cover of understory grass in southwestern (Arizona and New Mexico) ponderosa pine forests was highly dependent on the occurrence of monsoonal summer rain showers. He noted that, "Shallow-rooted herbs are dependent on summer showers, whereas the deeper rooted shrubs and trees are able to grow with little or no summer precipitation if the soil is well saturated during the winter months... A pronounced midsummer rainy season as well as relatively heavy soil favors the grasses."

Swetnam and Betancourt (1998) showed evidence from Arizona and New Mexico that fires in both mixed conifer forests and ponderosa pine forests in that region were most likely in years of low precipitation, but their analysis also suggested that fire in the latter forest type was positively correlated with precipitation in prior years. Swetnam and Betancourt (1998) ascribed this pattern to the increased importance of fine fuels – both live (grass) and dead (needles) – to fire dynamics in the more open and drier ponderosa pine stands, whereas moister mixed conifer forests are more characterized by woody fuels that build up gradually over years and are less responsive to annual changes in precipitation. California supports a Mediterranean climate which is similar to the southwestern climate (AZ and NM) in its overall aridity but very different in its distribution of rainfall. For example, July and August are the driest months in California, but the

wettest in Arizona and New Mexico; in much of New Mexico, the driest months are December through February, which is the height of the rainy season in California. These differences affect herbaceous vegetation much more profoundly than woody vegetation, and the general lack of rainfall during the growing season in the assessment area means that grass and forb production is severely water limited in upland sites.

Norman and Taylor (2003) carried out a study of fire history and fire-climate interactions in pine forests found along meadow edges on the Lassen National Forest in an area of relatively subdued topography. They found that the historic occurrence of widespread fires was partly dependent on moisture conditions 1-3 years prior, which suggested that grass production in and around the meadow systems was important to fire spread. It is difficult to extrapolate this study to the broader landscape, as most YPMC forests in the assessment area are not located adjacent to extensive meadow complexes, and grass cover in upland forests is generally not sufficient to carry fire on its own (Fites 1993, Smith 1994, Oliver 2000).

In summary, there appears to be little basis for the idea that presettlement YPMC forests in the assessment area supported abundant swards of grass in their understory, except possibly in areas of high soil moisture and relatively open canopy conditions, and at lower elevations in yellow pine dominated stands, especially where these interfinger with oak woodland. Except in these places, surface fires may have been primarily fed by tree litter, dried forbs, and shrubs, with a minor and variable component of grass. Variability in winter and spring precipitation certainly played a role in driving interannual differences in herbaceous and graminoid biomass however.

Although grass cover in presettlement YPMC forests may not have been particularly high, many forbs are more shade tolerant, and overall herbaceous cover and local species richness has probably dropped as tree cover has increased with fire suppression. Parks (2009) notes that stem densities of understory herbaceous species in the Stanislaus-Tuolumne Experimental Forest have been reduced by around 80% over the last eight decades as a result of increased in forest stand density and canopy cover.

Another wildcard is the increased presence of cheatgrass (*Bromus tectorum*) in YPMC forests in the assessment area. In eastside and lower elevation westside forests, this invasive grass can take advantage of disturbance to invade forest stands. Wet, warm years can increase cheatgrass cover substantially, and in recent wet years we have seen yellow pine stands with substantial cheatgrass cover in their understories where little understory existed before.

Future:

As with shrubs, it seems like two trends are likely. On the landscape scale, it seems inevitable that current and projected future trends will lead to increased importance of (early seral) herbaceous vegetation on the YPMC landscape. On the other hand, in undisturbed forest stands forest cover and density will continue to increase and herbaceous diversity and abundance will continue to be suppressed.

Litter

NRV and comparison to current:

There are no measurements of litter depth or cover that we know of from YPMC forests in the late 19th or early 20th centuries. However, many early observers remarked as to the common occurrence of bare soil and the prevalence of thin layers of needle litter on the soil surface (e.g., Fitch 1900, Sudworth 1900, Leiberg 1902).

Modern litter+duff depth measurements from contemporary reference sites tend to average between 1.5 and 2 cm (e.g., Lydersen and North [2012]: 2.1 cm [\pm 1.8 st. dev.]; Stephens [2004], Sierra de San Pedro Mártir: 1.6 cm (no st. dev. given); Safford, unpub. data, Sierra Juarez: 2.1 cm [\pm 1.6 st. dev.]).

The FIA data compilation (USFS 2013) for YPMC forests gives an average of 3.4 cm (\pm 4.5 st. dev.), with mixed conifer plots averaging 3.9 cm and yellow pine plots averaging 2.8.

These are very few data points to base an ecosystem-wide conclusion on. See the Forest Fuels section (below) for more evidence relating to forest floor accumulations of dead plant matter.

Forest fuels

NRV and comparison to current:

Stephens (2004), working in reference YPMC forests in the Sierra San Pedro Mártir in Baja California, found average surface fuel loads (1 hr to 1000 hr summed) of 15.8 tons/ha (\pm 3.9 SE; 1-100 hr sum was 2.16 tons/ha) and ground (litter+duff) fuels of 8.7 tons/ha (\pm 0.83 SE). Fire is frequent enough in this system (in interaction with the semiarid climate) to result in almost no duff (Stephens 2004). Fuel loadings were extremely variable. For example, surface fuel loads were below average on 73% of the plots, and above 36.8 tons/ha on 8% of plots. 37% of the plots sampled had no 1000 hr fuels (branches and logs >7.6 cm [3"] diameter), and 20% of the plots supported 75% of the 1000 hr fuels.

Lydersen and North (2012), in their study of old growth stands in the assessment area that had experienced at least two fires over the previous 65 years, found an average of 40.2 tons/ha for all surface fuels (1 hr to 1000 hr summed; 1-100 hr sum was 7.9 tons/ha). Variability was high, with the standard deviations for all of the component fuel types about equal to or greater than the mean. Fuel loadings were lowest on ridgetops and SW-facing slopes.

Taylor et al. (in press) used three methods to estimate pre-fire suppression fuel loads in reconstruction plots in YPMC forests in the Lake Tahoe Basin (see paper for detail). Average estimated fuel loads from the three techniques ranged from 4.2-6.1 tons/ha (1 hr to 100 hr summed) for Jeffrey pine forest and 4.4-8.3 tons/ha for mixed conifer forest. This compared to contemporary ranges of 4.2-7.5 tons/ha for Jeffrey pine forest and 8.5-12.9 tons/ha for mixed conifer forest.

Studies of prescribed fire in modern, fire-suppressed forest can provide insight into fuel loadings that may have characterized presettlement YPMC forests in the assessment area. Kauffman and Martin (1989) report results from 60 prescribed fires from three sites in the northern Sierra

Nevada. We report only their early fall and late spring results (30 burns), as those fires are most likely to represent conditions like those during the natural fire season. After fire, Kauffman and Martin (1989) found an average of 1.5 tons/ha (range of site means 0.6 to 2.2) for 1 hr to 100 hr fuels, and an average of 12.8 tons/ha (5.8 to 18.7) for 1 hr to 1000 hr summed fuels; ground fuels (litter+duff) averaged 11.6 tons/ha (Figure 21). Unburned forest supported over eight times more ground fuels by mass, more than five times more fuel in the 1 to 100 hr classes, and 2.8 times more fuel when the 1 to 1000 hr classes are summed (Figure 21). Other studies of prescribed fire effects on fuels provide remarkably similar results. The means of postfire 1-100 hr fuels, 1-1000 hr fuels and ground fuels from Kilgore and Sando (1975), Stephens and Finney (2002) and Knapp et al. (2005; fall burn) are 1.4 tons/ha, 12.5 tons/ha, and 10.7 tons/ha. Average prefire fuels in these studies was higher than in the Kauffman and Martin study sites. We would expect average presettlement fuel loadings to be less than the postfire loadings reported here, as these modern prescribed fires were one-time events after many decades of fire exclusion, whereas presettlement YPMC forests were experiencing burns every 10-20 years on average.

The FIA plot data compilation (USFS 2013) shows that fuel loadings in contemporary YPMC forest plots average 7.3 tons/ha for 1-100 hr summed fuels, and 30.3 tons/ha (+/- 24.9 st. dev.) for 1-1000 hr fuels. Mixed conifer forests support much heavier fuels (1-1000 hr summed = 36.7 tons/ha) than yellow pine forests (1-1000 hr summed = 20 tons/ha). This gives a good idea of the kind of transformation that occurs in the fuel profile of YPMC forests as they transition from pine dominated stands to mixed conifer stands under fire suppression. The average of the reference studies detailed above is about 3.6 tons/ha for 1-100 hr fuels, and 17.7 tons/ha for 1-1000 hr fuels (the latter including the very high value from Lydersen and North [2012]). We conclude that modern-day fuel loadings in assessment area YPMC forests are substantially higher than loadings under presettlement conditions.

Future:

We know of no scientific attempt to project fuel levels in assessment area YPMC forests into the future. It seems most likely that the same sorts of trends that have characterized YPMC forests for the last 50-100 years will continue into the future, perhaps at an accelerated pace, as climate warming and rising CO₂ levels increase plant growth and fuels accumulation.

Composition

Forest landscape composition

NRV and comparison to current:

Here we refer to the distribution of forest types (rather than specific species) across the landscape. There are not many data available, as mapping or extensive plot networks are necessary in order to assess this indicator.

Many early observers noted how yellow pine-dominated forests (yellow pine and dry mixed conifer) were more common at lower elevations, on warm aspects (south and west), and in areas of thin or otherwise low productivity soils. Forests with a notable shade tolerant component (moist mixed conifer) were mostly in moist microsites, along streams, on north slopes, and at high elevations (e.g., Sudworth 1900, Leiberg 1902). These general trends are also apparent today, although the absolute area supporting yellow pine-dominated forest has dropped and the area supporting shade tolerant-dominated forest types has increased (Vankat and Major 1978, Barbour et al. 1993, 2007; Fites-Kaufman 1997, Sugihara et al. 2006)

Fites-Kaufman (1997) developed environmental models of presettlement forest types in two watersheds on the western slope of the central assessment area. She concluded that 64% of the areas modeled would be dominated by Douglas-fir and white fir forest types in the absence of disturbance (i.e., on cool and moderate slopes, watered draws on warm slopes, and higher elevations), but 28% of that area was in intermediate areas likely historically dominated by ponderosa and sugar pine due to frequent fire. Overall, according to Fites-Kaufman (1997), about 2/3 of the area modeled was probably dominated by pine-dominated forest before Euroamerican settlement.

Dolanc et al. (in prep.), compared the vegetation types represented by 4371 VTM forest plots (Wieslander 1935) from the 1930s and 1000 FIA forest plots from the 2000s in the central assessment area. They found that 19.7% of the 1930s plots classified as ponderosa pine, versus 8.9% of the plots from the 2000s. 27.4% of the plots classified as mixed conifer in the 1930s dataset, versus 37.1% in the 2000s dataset. Both eastside and westside Jeffrey pine also declined as a proportion of the sampled vegetation between the 1930s and 2000s. Dolanc et al. (in prep.) suggested that much of the ponderosa pine sampled in the 1930s and subsequently lost probably succeeded to mixed conifer (through infilling of fir, Douglas-fir and incense cedar) or to montane hardwood forest, through expansion of oaks after disturbance.

Thorne et al. (2008) compared the 1930s Forest Service vegetation maps (Wieslander 1935) of the Placerville Quadrangle, in the center west of the assessment area, with the Forest Service vegetation map from 1996, in order to compare the extent of different vegetation types in the two time periods. The vegetation type with the largest loss of area was ponderosa pine, which declined by 64% over the six decades between maps. In their study area, Thorne et al. (2008) documented that ponderosa pine forest had primarily transitioned to forests dominated by hardwoods (where disturbance had reduced or removed the dominant pines) or by Douglas-fir (where lack of disturbance allowed succession of shade tolerant species). Thorne et al. (2008) did not find much change in their mixed conifer forest type, but their classification (California

Wildlife Habitat Relations – CWHR) splits a number of forest types we include in our general definition of YPMC forests.

Current Forest Service vegetation maps can also be compared to Show and Kotok's (1929) summary of forest cover types in northern California in the late 1920s. The major changes are in yellow pine and mixed conifer. The former comprised 33.7% of Show and Kotok's analysis area in the 1920s (Modoc National Forest south to the Sequoia National Forest, excluding the Inyo National Forest), but only 17% of the area in the most recent Forest Service mapping. Mixed conifer forests covered 19.8% of the area in the 1920s, versus about 30% today. Other forest types were similar between the two time periods (current data courtesy of Jay Miller, USFS).

Future:

See following section.

Forest species composition and species diversity

Trees

A number of excellent summaries of the species composition of assessment area YPMC forests have been published (e.g., Fites 1993, Smith 1997, Barbour and Billings 1999, USDA 2001, Sugihara et al. 2006, Fites-Kaufman et al. 2007). We direct the reader to these and other sources for information on modern forest conditions. In this section we both qualitatively and quantitatively describe the patterns of tree composition that characterized YPMC forests before significant Euroamerican impact.

NRV:

The US Geological Survey carried out general surveys of Sierra Nevada forests at the turn of the 19th century, primarily to assess the status of the timber resource. Leiberg (1902) surveyed forestlands on what is today the Plumas and Tahoe National Forests and the northern Lake Tahoe Basin. Sudworth (1900) surveyed the southern Lake Tahoe Basin, the Eldorado National Forest, and northern portions of the Toiyabe and Stanislaus National Forests. Fitch (1900) and Marshall (1900) provided brief descriptions of forest conditions in and around Yosemite National Park. McKelvey and Johnston (1992) evaluated the USGS reports and provided a summary of their results. Stephens and Elliot-Fisk (1998) and Stephens (2000) summarized unpublished plot data from Sudworth that were entered into his field notebooks, including data from the southern Sierra Nevada in what is today Sequoia-Kings Canyon National Parks and the Sierra and Sequoia National Forests.

McKelvey and Johnston (1992) note that the composition of forest trees in these early surveys included all of the same species we encounter today, but the order of their dominance in assessment-area forest stands has changed. Leiberg (1902) divided his survey area into three predominant forest types: his "yellow pine" forest type includes what we call today "mixed conifer" and corresponds to the YPMC forest types covered in this chapter. According to Leiberg, yellow pine was the "most conspicuous and important" species in this forest type, but although it may have once been the dominant species of tree, due to heavy selective logging throughout his

survey area it was no longer so by the turn of the century. Sugar pine (*Pinus lambertiana*) was not a dominant species in YPMC forests, but like the yellow pine species, it was selectively harvested and its densities and overall volume had decreased greatly by the time the USGS surveys were made. Leiberg's (1902) opinions are qualitatively supported by the forest observations of Joseph Bruff, who walked the same northern Sierra Nevada forests a half century earlier. Bruff occasionally made journal notes of forest structure and composition in his 1849-1851 travels, and almost all of his observations of conifer trees refer to forests dominated by pines (Read and Gaines 1949).

Sudworth (1900) stated that pine species dominated lower elevation YPMC forests, but incense cedar and fir were mixed with the pine at approximately equal proportion at the higher elevations. Sudworth (1900) rarely referred to hardwood species, but he did note that "a few small, unimportant broad-leafed trees" were common along canyon-bottom streams and also grew at times in with the upland coniferous forest. According to Sudworth, yellow pine was the most abundant tree in the YPMC belt, followed by white fir, then incense cedar, sugar pine, and Jeffrey pine. The amount of yellow pine varied considerably from place to place and along the altitudinal gradient, it was most common on south, west, and east aspects, with some sites containing 80-90% yellow pine. Most forest stands included different mixes of the above species, with Sudworth estimating the typical ranges being: 45-50% yellow pine; 30-45% white fir; 20-30% incense cedar; 5-20% sugar pine; 0-5% Jeffrey pine on the west slope, but a local dominant on the east slope; Douglas-fir 2-5%; Black oak was 5-10% of stands at lower elevations. Black oak was a close associate of yellow pine and incense cedar and could account for more than 50% of a stand in dry, thinly soiled locations on south and west slopes.

Sudworth and the other USGS surveyors were visiting forests that had already had nearly a half century of Euroamerican presence. Many of the stands they surveyed had been cut, usually selectively for pine. This had an impact on their estimates of species importance. For example, Sudworth (1900) noted that the best sugar pine had already been cut out from much of the forest he surveyed, and the same had happened with the larger specimens of Douglas-fir. Leiberg (1902) described the effects of heavy logging on forest in the Lake Tahoe Basin and noted that sugar pine, which had comprised 20-25% of the virgin stand, was likely to contribute only 2-3% of the secondary forest. Greeley (1907) lamented that many stands had been so thoroughly high-graded (valuable species and large trees selectively cut) that yellow pine averaged only 15-20% of the trees in much of the western Sierra. Leiberg and other early surveyors also noted that the contribution of incense cedar to the stands they visited was artificially high, as it had no commercial value and was left standing far more often than the other species.

Leiberg (1902) estimated that YPMC forests in his survey area at the time of his work were on average 25% yellow pine, 1% sugar pine, 25% Douglas-fir, 44% white fir, and 5% incense cedar. The relative proportions of the pines and fir had been markedly changed by logging, which focused heavily on the former. At the time of the USGS surveys, Leiberg's survey area had been (much) more heavily cut than the area assessed by Sudworth. Leiberg (1902) also estimated the standing volume of trees species in as yet uncut forests. Proportionally, yellow pine species (both ponderosa and Jeffrey) were dominant, and comprised about twice the volume of Douglas-fir, sugar pine, or white fir (see McKelvey and Johnston 1992); incense cedar had no economic value at the time and was not listed. White fir regenerated strongly on formerly

pine-dominated lands and Leiberg estimated that it would amount to 60-75% of the secondary forest, up from 25-40% of the uncut stands. Sudworth (1900) also referred to the “thickets of seedlings and saplings” of white fir that covered many acres and often excluded all other species. Other observers made similar observations.

Volume estimates from the Plumas National Forest in 1913 are very similar to Leiberg’s earlier volume estimates for the northern Sierra Nevada as a whole. As above, yellow pine species dominated, with 1.6 times the volume of white fir, 2.4 times the volume of sugar pine, and 2.8 times the Douglas-fir volume; other species like red fir and incense cedar comprised the remaining 7% of volume (Moore 1913, McKelvey and Johnston 1992).

W.B. Greeley (1907) described the forests of the western Sierra Nevada from the viewpoint of a traditional forester. Like the other early observers, he noted that the original forest had varied from pure yellow pine at the lower elevations and on dry exposures to nearly pure fir toward the upper limit of merchantable timber. Greeley outlined “three bad features of the virgin stand (that) quickly impress the forester.” Two of these three features pertain to structure (“the widespread over-maturity of the timber” and “the large area on which the stand is open or has disappeared entirely”) and are discussed above. The third feature was the “large and apparently increasing proportion of inferior species.” Except at lower elevations, cutting had been sufficient to reduce the pines to where white fir comprised a third of merchantable timber, and it was a much higher proportion of the younger growth. Together with incense cedar, white fir outnumbered both of the pines in small size classes. Both white fir and incense cedar produce high volumes of seed, and their seedlings are much more tolerant of shade than the pines. Greeley (1907) noted that, except at lower elevations and on very warm exposures, dense thickets of fir and cedar were already crowding the pine to “one fifth or less of reproduction”.

Early General Land Office (“GLO”) land surveying of the assessment area produced useful data on the composition of late 19th century forests. GLO crews sampled up to four trees at each section corner and two trees at quarter-section corners (see above). Because of the gridded sampling scheme, GLO samples of forest structure and composition are relatively robust, statistically speaking, although they did not restrict their activities to uncut ground and they tended to prefer smaller mature trees when they were available, as they were assumed to have the greatest longevity (Hyde 2002). The very low density of GLO data (a maximum of eight points per square mile) means that spatial interpretation of these data is nearly impossible (not to say that people haven’t tried), but they are very valuable as tabular summaries from larger landscapes. GLO tree data have been summarized for the Lake Tahoe Basin (Manley et al. 2000), the Eldorado National Forest (Fites-Kaufman 1997), and for the Stanislaus, Sierra, and Sequoia National Forests (Hyde 2002).

GLO data from Lake Tahoe suggest that yellow pine and fir were present in nearly equal numbers in lower montane forests, with the balance shifting from fir on the wetter west shore to pine on the drier east shore (Manley et al. 2000). Many of the GLO surveys in the Tahoe Basin were carried out after the mid-1870s however, and in the lower montane zone their section lines, especially on the east shore, crossed stands where selective logging of Jeffrey and sugar pine had already commenced (Lindström 2000). In addition, the data summarized in Manley et al. (2000) are missing many survey points. It seems likely that the GLO counts in the Lake Tahoe

Basin in lower montane forests may underestimate the presence of pine before logging, at least on the east shore. Data from Taylor's studies in the Tahoe Basin support this viewpoint (Taylor 2004, Taylor et al. in press). For example, the modern Jeffrey pine stands sampled by Taylor and colleagues on the east shore of the lake support an order of magnitude more Jeffrey pine than white fir, even a century after the institution of fire suppression and 130 years or more after the removal of all merchantable Jeffrey pine. Taylor et al.'s (in press) results from west shore mixed conifer forest agree more closely with the GLO data, with white fir in the 1873 forest outnumbering Jeffrey pine by 1.5:1 (vs. 5:4 in the GLO surveys). Almost half of all of the white fir measured were smaller than 25 cm dbh, and Jeffrey pine was the dominant tree in the larger size classes (Taylor et al., in press).

Hyde (2002) summarized compositional patterns in GLO data sampled in the middle and late 1800s from areas on the Stanislaus (37300 ha sampled), Sierra (46600 ha), and Sequoia National Forests (18600 ha). Her study areas were chosen to represent elevational transects of approximately 2000 m, along a latitudinal gradient from the central to southern Sierra Nevada. Below 1000 m elevation, forest vegetation was dominated by oak species (more than $\frac{3}{4}$ of stems sampled), with only about 8.5% of the trees measured being pines (although these contributed about $\frac{1}{4}$ of the basal area). Between 1000 and 1500 m elevation, pine (45% of sampled stems) and oak (37% of sampled stems, mostly black oak) shared in dominance, with fir species contributing 8% of stems. Pines comprised about 60% of the basal area, and oaks about 24% (Hyde 2002). Between 1500-2000 m elevation, pines (mostly yellow pines and sugar pine) accounted for 58% of stems, fir about 26%, and oaks about 11%. Basal area was dominated by pines (49%), giant sequoia (29%) and white fir (16%). Above 2000 m, the fir component increased to 42% of stems sampled, with pines comprising about 57%; yellow pine, lodgepole pine and sugar pine were the dominant pine species, in that order. Fir (probably mostly red fir at these elevations) contributed slightly more basal area to the stand than the pines (Hyde 2002). Pines dominated slopes of all aspects except northeast, which was dominated by oak species; fir species were surprisingly constant in their contribution to stem densities, comprising in all cases about 20% of stems (Hyde 2002).

Fites-Kaufman (1997) summarized the GLO data for two watersheds on the Eldorado National Forest. She found that GLO sites recorded "fir" (Douglas-fir or white fir) on only 26% of the surveyed sites. Although Fites-Kaufman (1997) does not report the overall results of the GLO data, her Figure 2.11 gives the proportion of stems by species for GLO sites that contained "fir" and for sites that her modeling had identified as being within the Douglas-fir-mixed conifer potential forest type (i.e., on cool slopes, higher elevations, etc.). Ponderosa and sugar pine accounted for 24% and 31% of the stems in the two subsamples, black oak accounted for 26% and 30%. These fire tolerant species accounted for much more of the stands sampled by the GLO across the rest of the landscape, but Fites-Kaufman (1997) does not provide a summary of those data.

The Forest Service inventoried and mapped vegetation on much of the land under its jurisdiction in the early 1930s ("VTM" project; Wieslander 1935). In the assessment area, thousands of 1/5-acre (809 m²) vegetation plots were sampled in YPMC forest types (ponderosa pine, mixed conifer, and eastside Jeffrey pine forests). Figure 22 gives the relative densities of tree species in YPMC forests sampled by the VTM project in forest stands that had not experienced intensive logging (n = 2250), and compares them to the most recent available FIA inventory of assessment

area forests ($n = 510$; Dolanc, et al., in prep.). The VTM protocol only sampled trees 4" (10 cm) dbh and above, so the relative densities of species sampled are biased away from the youngest cohorts of trees, which were dominated by white fir and incense cedar that were beginning to benefit from two to three decades of fire suppression. Because of this, relative tree densities represented in the VTM data set provide an at least approximate idea of dominance patterns among tree species before fire suppression (with the caveat that large and medium-sized sugar pine and yellow pine had been selectively logged through much of the assessment area by the time of VTM sampling). The values given in Figure 22 are overall averages, and obviously patterns varied from westside to eastside forests (e.g., Jeffrey pine strongly dominated eastside stands, and Douglas-fir and black oak were much more common on the westside), from north to south slopes, and so on. The most obvious pattern in Figure 22 is the strong shift in overall dominance from shade intolerant/fire tolerant species (ponderosa pine, Jeffrey pine, sugar pine, black oak) in the VTM data set to shade tolerant/fire intolerant species (white and red fir, incense cedar, Douglas-fir) in the FIA data set. The relative proportions of shade intolerant vs. shade tolerant species change from 60:40 in the VTM data, to 35:55 in the FIA data set (Figure 22; Dolanc et al., in prep.).

Figure 23 shows historical stem densities by species for seven sites in the assessment area where stand reconstruction studies were carried out. Ratios of shade intolerant to shade tolerant species range from 92:7 to 16:84 (Figure 23). It can be appreciated that the relative densities of shade intolerant and shade tolerant species were not constant on the landscape, and in some circumstances yellow pines, which dominated much of the general landscape, were entirely absent.

In another forest reconstruction study based on "aggregations" of vegetation, Bonnicksen and Stone (1982) found that a YPMC watershed in Kings Canyon National Park had seen an increase in the area of the watershed covered by white fir-dominated aggregations from 27% to 37% between 1890 and 1977. Black oak was reduced from 10% of the study area to 6% during the same period.

Kercher and Axelrod (1984) developed a Monte Carlo-based model of YPMC forest succession ("SILVA") in the Sierra Nevada. This model is further discussed in the Forest Successional Processes section. Kercher and Axelrod (1984) used SILVA to compare forest succession after a simulated clear-cut for 500 years at two different elevations, 1520 m (5000 ft) and 1830 m (6000 ft). The lower elevation site is at the upper reaches of YPMC forests historically dominated by ponderosa pine, the upper site is nearer the upper limits of YPMC forests in the Sierra Nevada and historically included a significant component of fir species. The time averaged results of the SILVA lower elevation simulation under frequent fire (mean fire return interval = 7 years) are given in Figure 11. Ponderosa pine strongly dominated the modeled stand throughout the 500 year simulation. The relative proportions of shade intolerant vs. shade tolerant species in Figure 11 are about 66:33. The SILVA model for the higher elevation site projected a forest stand dominated strongly by white fir (which comprised an average of 60% of the total basal area over the 500 year simulation), even under frequent fire (Kercher and Axelrod 1984).

In summary, presettlement YPMC forests in the assessment area were dominated by yellow pine and other shade-intolerant species, but shade-tolerant species could be locally dominant,

especially at higher elevations and in moist microsites. Most assessments of species densities were carried out after selective logging of yellow pine and sugar pine had already occurred. The actual proportions of shade tolerant to shade intolerant species varied across the presettlement landscape, but the data suggest that among mature trees, shade intolerant (pine and black oak) species typically comprised most of the individuals in the average stand, especially the larger individuals (see below). Then, as now, in the absence of fire, shade tolerant species dominated the smallest size classes.

Comparison with current:

Yellow pine and sugar pine have notably decreased in importance in assessment area YPMC forests, while shade tolerant conifers and hardwoods (white fir, incense cedar, Douglas-fir, tan oak, canyon live oak) have increased (Figure 24). This is due to selective logging, increased resource competition under increasing stand densities, the difficulty in regenerating under the denser modern canopy, and – in the case of sugar pine – to the effects of White Pine Blister Rust (*Cronartium ribicola*), an introduced fungal pathogen that is having major effects on 5-needle pine populations across the western US (Agee 1993, Barbour et al. 1993, 2007; van Mantgem et al. 2004, Sugihara et al. 2006). FIA data show that the component of shade intolerant species in assessment area YPMC forests has dropped from an average of maybe 2/3 of the mature forest stand to around 1/3 of mature trees. The largest decrease has been in ponderosa pine, but all shade intolerant species have decreased in density.

Geographic, topographic, and successional patterns in tree species composition are driven by ecological differences between species. Table 2 (intro) lists the tolerances of common YPMC tree species in the assessment area to shade, frost, temperature, drought and fire. The dominance of the yellow pine species on south and west aspects, on droughtier soils, and in fire-prone locations is easy to predict from the information in Table 2, as is the heightened presence of white fir on north slopes, and in moist and protected topographic locations; incense cedar is somewhat intermediate in most characteristics. White fir is the major successional climax species for most of the YPMC belt in the assessment area, although Douglas-fir plays this role at lower elevations. White fir's shade tolerance and fire intolerance, its ability to survive for very long periods as a suppressed tree, and its capacity to respond rapidly to release (increased light availability) are dominant factors in forest succession in the assessment area (Burns and Honkala 1990, Barbour et al. 2007, FEIS 2013). In pre-fire suppression times, very frequent fire greatly limited white fir populations in most of the assessment area, but it could dominate stands in areas where conditions existed (such as high fuel moisture) such that fire frequency was reduced. Increasing annual precipitation and lower summer temperatures along the south-to-north axis of the assessment area also result in higher fir densities to the north. The same gradient occurs with elevation, and many early observers noted how YPMC forests were dominated by white fir at higher elevations. With the onset of fire suppression, major increases in white fir and incense cedar began across the assessment area (Vankat and Major 1978). Early observers noted this tendency already in the late 19th and early 20th century (Sudworth 1900, Leiberg 1902, Greeley 1907). Today, with the general absence of fire, climate is the principal factor regulating species distributions, and species requiring frequent disturbance to retain their place in the forest

canopy – principally species of pine – are in the decline, except in areas of low site productivity such as areas of thin or nutrient-poor soil or extreme microclimates.

Dolanc et al. (in review) compared forest conditions as documented in the Forest Service forest mapping inventory of the early 1930s (Wieslander 1935) with the most recent compilation of the FIA (Forest Inventory and Analysis) plot network in the assessment area. Figure 24, taken from Dolanc et al. (in review) is a graphic summary of the relative changes in tree composition that have occurred in the assessment area since the arrival of Euroamericans. The YPMC belt begins where PIPO (ponderosa pine) is located on the X-axis and continues to PIJE (Jeffrey pine). The most obvious pattern in Figure 24 is the reduced frequency (calculated as the percent of plots in which the species in question is found) of shade intolerant/fire tolerant species like black oak (QUKE) and ponderosa, sugar (PILA) and Jeffrey pine; and the increased frequency of shade tolerant/fire intolerant species like Douglas-fir (PSME), CADE (incense cedar), and white fir (ABCO). Changes have been driven primarily by fire suppression and timber harvest.

Future:

Miller and Urban (1999b) added a soil water routine and a fire model to the forest gap model ZELIG and simulated successional dynamics in Sierra Nevada forests along an elevational gradient in Sequoia-Kings Canyon National Parks under a number of future climate scenarios. Simulations were run for 800 years, with the first 200 with no fire and the last 600 under a “natural” fire regime for the elevation in question. In all future climate scenarios except the cool-wet scenario, tree species composition at the two lowest elevation sites (1800 m and 2200 m), which are at elevations currently dominated by YPMC forests, changed markedly. Major changes were not apparent in the models until about a century into the simulations, but fire in Miller and Urban’s (1999b) model is entirely surface fire and is internally driven (by fuel load and fuel moisture), not externally forced, and effects of disease or insect outbreak were not modeled, so the results are probably very conservative with respect to the velocity of change. At the lower two elevations, trees either completely or nearly disappeared by the final century of the simulation. Under the less extreme warming and drying scenarios, the forest at 1800 m saw the basal area of tree species change from 15:9:3:1:1 (m²/ha; ponderosa pine:incense cedar:Jeffrey pine:black oak:white fir) to 1:1:1:1:0. Under more extreme warming, trees disappeared at this elevation. At 2200 m, baseline basal areas of 45:4:2:1:0 (m²/ha; white fir:sugar pine:incense cedar:Jeffrey pine:ponderosa pine) were changed to an average of 4:0:12:2.5:10 under the less extreme warming/drying scenarios, and the site transitioned to a yellow pine-black oak-incense cedar woodland (total basal area only 8 m²/ha) under the more extreme scenarios. The 2600 m site, which currently supports a red fir forest, transitioned to a moist mixed conifer forest dominated by white fir under the less extreme future scenarios, and to a dry mixed conifer (yellow pine-white fir-incense cedar in almost equal proportions) under the more extreme scenarios (Miller and Urban 1999b).

The Nature Conservancy carried out future climate suitability forecasts for major tree and shrub species of the Sierra Nevada, in support of the northern and southern Sierra Nevada Partnerships. The Partnerships are collaborative efforts, organized by TNC, the Sierra Business Council, and a variety of other national, regional and local conservation organizations, to carry out all-lands conservation planning for important subregions of the Sierra Nevada (see: <http://consbio>).

[org/products/projects/southern-sierra-partnership](http://products/projects/southern-sierra-partnership), and <http://northern-sierrapartnership.org/>). Climate suitability forecasts for the period 2045-2065 were derived using maximum entropy (MaxEnt) modeling, and based on 11 GCMs run under the IPCC's A2 climate scenario. Models only incorporated climate variables. Three generalized future climate scenarios were developed ("warm-dry", "hot-dry", and "hot-wet") and an ensemble projection was also developed, where the degree of consensus across models was mapped. In the ensemble outputs, areas of projected climate "stress" (possible loss in distribution), climate refugia, and possible expansion were mapped, along with a measure of the degree of model consensus. More information and the data outputs are available at <http://app.databasin.org/app/pages/galleryPage.jsp?id=8c5db744f9fe4d3e9375b100dc695c4d>. All of the projections for California can be viewed at <http://www.tnccmaps.org/climate/species/>.

Overall, the TNC climate suitability forecasts suggest more climatic stability and less future climate stress in the southern Sierra Nevada than in the north (Southern Sierra Partnership 2010). This is due largely to much higher elevations and more accentuated topography in the south, which lead to less probability of overall habitat loss as climates warm and snowpack decreases. Nonetheless, the models suggest notable areas of climate stress along the lower, western margins of the YPMC forest belt, as well as the potential for expansion to higher elevations (Figure 25). Both Jeffrey and ponderosa pine are projected to experience increased climate stress in much of the northern and northeastern assessment areas, primarily because of the lower average elevations in those areas. Douglas-fir (not shown) is projected to experience relatively little climate stress, except along the western margins of the central Sierra Nevada, and climate suitability for the species may increase markedly at higher elevations throughout the central and northern Sierra Nevada. In their summary of the top threats to conservation in mixed conifer forests, the Southern Sierra Partnership (2010) identified changed fire regimes as the highest threat, followed by climate change and a number of other factors.

Other plant life forms

NRV and comparison to current:

Early observers did not measure richness of diversity of species other than trees, although they made reference to certain shrubs (mostly those that competed with trees), and sometimes referred in passing to herbaceous species as well. For example, Muir (1894) noted how the open canopy of YPMC forests in the assessment area led to many understory species, especially shrubs and forbs, but the growth was "never so dense... as to prevent the traveler from sauntering where he will" (Muir 1984, Chapter 3). A general ecological principle is that the diversity of understory species tends to be negatively correlated with overstory canopy cover, and positively correlated with light availability (Barbour et al. 1987). Presettlement YPMC forests were generally more open than modern forests, and the enhanced incidence of light at the forest floor likely led to higher local species richness in understory plants, but we can only theorize based on numerous modern demonstrations of the negative relationship between forest canopy cover and understory species diversity in forests in and around the assessment area (e.g., Agee and Biswell 1970, Battles et al. 2001, Keeley et al. 2003, Wayman and North 2007, Webster and Halpern 2010, Kuhn et al. 2011). These probable local effects notwithstanding, there is no evidence that we know of that demonstrates any notable change in regional richness (overall species diversity

across the assessment area) of understory plants in the assessment area since presettlement times.

Although many plant species in California have been demonstrated to require fire scarification of their seeds to induce germination (e.g., through heat, presence of certain volatiles, or chemicals contained in charcoal), such plants are rare in YPMC forests in the assessment area (Webster and Halpern 2010; Keeley et al. 2003, 2012). This is probably due to the relative rarity of high intensity fire over evolutionary time in these forests (Grubb 1977, Denslow 1980). As such, the large decrease in fire frequency and overall area burned in assessment area YPMC forests since presettlement times has probably not had a highly deleterious effect on regional understory diversity, or at least as deleterious an effect as it might theoretically have in an ecosystem characterized by highly intense fire. Several widely distributed shrub species from the genera *Ceanothus* and *Arctostaphylos* do possess fire-promoted seed germination, and although these are typically shaded out by overstory trees over time, their soil seedbanks may persist for hundreds of years and they are not at regional risk of extinction due to fire suppression (Keeley et al. 2012).

As with the canopy tree species, understory species composition varies across the assessment area, depending on underlying environmental conditions as well as the influence of the tree canopy layer. It seems probable that understory species requiring higher light environments may have become less abundant (if not less species rich) in the assessment area as a result of fire suppression, but there is no direct evidence to base such a conclusion on.

Future:

The Nature Conservancy forecasted future climate suitability for a number of YPMC forest shrub species using MaxEnt modeling. See the Forest Composition and Species Diversity:Trees section for more detail on the modeling. Outputs are viewable at <http://www.tnccmaps.org/climate/species/>. These models suggest notable loss of suitable climates for important species like Greenleaf manzanita (*Arctostaphylos patula*) and mountain whitethorn (*Ceanothus cordulatus*) in the northern and northeastern assessment areas. As noted above, these models include only climate variables and many other factors interact to drive species distributions.

See the Forest Structure:Forest Understory section for more information on understory vegetation.

SUMMARY OF PROBABLE DEVIATIONS FROM NRV

Based on our understanding of yellow pine and mixed conifer ecosystems in the bioregional assessment area, we have attempted to draw conclusions with regard to whether key ecosystem variables are currently within or outside of the natural range of variation (NRV). Table 11 summarizes our conclusions and directs the reader to the areas of this report that discuss the ecosystem elements in question.

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TABLES

Table 1. Dominant tree species of YPMC forests in the assessment area, their geographic distribution, approximate elevational range within the assessment area, and comparisons of leaf flammability and seed weight.

Species	Scientific name	Geographic distribution	Elevation (m) ¹ *	Leaf flammability ²	Seed weight (g) ³
Ponderosa pine	<i>Pinus ponderosa</i>	W. US, SW Canada, N. Mexico	300-1800 N; 1200-2100 S	Very high	0.02 - 0.07
Jeffrey pine	<i>Pinus jeffreyi</i>	E. edge of North American Mediterranean zone, S. Oregon to Baja California	1500-2400 N; 1700-2800 S	Very high	0.08 - 0.2
Sugar pine	<i>Pinus lambertiana</i>	Oregon to Baja California	1000-2000 N; 1400-2700 S	High	0.15 - 0.3
Incense cedar	<i>Calocedrus decurrens</i>	Oregon to Baja California	600-2100	Moderate	0.015 - 0.07
Douglas-fir	<i>Pseudotsuga menziesii</i>	W. US, SW Canada, N. Mexico	300-2100 N; 600-2100 S	Low	0.01 - 0.02
White fir	<i>Abies concolor</i>	S. Rockies, SW US, S. Oregon to Baja California	800-2300 N; 1500-2400 S	Moderate	0.015 - 0.055
Black oak	<i>Quercus kelloggii</i>	S. Oregon to Baja California	900-1500 N; 1400-2100 S	Very high	3 - 9

* Elevations rounded to nearest 100 m; N = north, S = south

¹ Sources: Storer and Usinger 1963, Lanner 1999, Fites-Kaufman et al. 2006

² Sources: Fonda et al. 1998, Engber and Varner 2012, Magalhães and Schwilk 2012

³ Sources: Fowells and Schubert 1956, Burns and Honkala 1990, Bonner and Karrfalt 2008

Table 2. Comparative ecological tolerances of common tree species in assessment area YPMC forests. Species arranged from low tolerance (top) to high tolerance (bottom). Data from Minore (1979), Burns and Honkala (1990), and FEIS (2013).

Shade	Frost	Temperature ¹	Drought	Fire ²
black oak/western juniper	madrone	lodgepole pine	red fir	lodgepole pine
ponderosa pine	Douglas-fir	red fir	white fir	sugar pine/white fir
lodgepole pine	white fir	Jeffrey pine	western white pine	incense cedar
sugar pine	sugar pine	white fir	sugar pine	Jeffrey pine/ ponderosa pine/ Douglas-fir
incense cedar/ western white pine	incense cedar	Douglas-fir/sugar pine/incense cedar	Douglas-fir	
Douglas-fir	ponderosa pine/ Jeffrey pine/red fir	ponderosa pine/ black oak/madrone	lodgepole pine/ incense cedar/ madrone	
red fir	lodgepole pine, western white pine		ponderosa pine	
white fir			Jeffrey pine	
			black oak	

¹ least heat tolerant/most cold tolerant on top

² fire tolerance of mature trees. Fir and Douglas-fir seedlings and saplings are less tolerant of fire than yellow pine and sugar pine

Table 3. Mean “full-grown” tree sizes (height and diameter at breast height - dbh) for major tree species in uncut YPMC forests in the central Sierra Nevada at the turn of 19th century, according to Sudworth (1900)

Species	Height (feet)	Height (m)	dbh (feet)	dbh (cm)
Yellow pine	150-180	45-55	3-4	90-120
Sugar pine	180-200	55-61	4-6	120-180
Incense cedar	80-100	24-31	5-7	150-210
White fir	175-190	53-58	3.5-5.5	110-170
Jeffrey pine	125-160	38-49	3-5	90-150
Douglas-fir	150-175	46-53	4-7	120-210

Table 4. Tree cover (%) in historical and contemporary reference yellow pine and mixed conifer stands in the assessment area and comparable reference sites. Sites arranged from north to south.

Site	Mean	St. Err.	Method	Observations	Source
Lassen National Forest	25.2 (29)*		FVS modeled estimate ¹	Modern: ponderosa pine forest, Ishi Wilderness	Taylor 2010
North and central assessment area	48.5 (79.9)*	1.2	FVS modeled estimate ¹	1930s: Forest Service vegetation mapping inventory, YPMC	Dolanc et al., in review
Central assessment area	69.1 (89.1)*	4.1	FVS modeled estimate ¹	1899: timber inventory, mixed conifer	Sudworth 1900
Lake Tahoe Basin	16.6 (18.1)*		FVS modeled estimate ¹	1870-1900: Stump counts, Jeffrey pine forest, east shore	Taylor 2004
Lake Tahoe Basin	35.7 (44.1)*		FVS modeled estimate ¹	1893: Plot-based reconstruction, mixed conifer, west shore	Taylor et al., in press
Lake Tahoe Basin	45.3		Line intercept	Modern: Uncut old-growth, YPMC, white fir, red fir	Barbour et al. 2002
Stanislaus National Forest	36/45**		Tree canopy area equations	1889/1929: Plot-based reconstruction, mixed conifer	Lydersen et al., in press
Yosemite National Park	22		FVS modeled estimate	1911: timber inventory, mixed conifer	Collins et al. 2011
Yosemite National Park	41 (52.7)*		FVS modeled estimate ¹	1899: Plot-based reconstruction, moist mixed conifer	Scholl and Taylor 2010
Sierra National Forest	33.9 (41.4)*		FVS modeled estimate ¹	1865: Plot-based reconstruction, moist mixed conifer, Teakettle Exp. Forest	North et al. 2007
Assessment area (mostly southern and central)	43.7	1.5	Densitometer	Modern: YPMC forest, uncut, with frequent and/or recent non stand-replacing fire	Lydersen and North 2012
Sierra de San Pedro Mártir	25.3	3.8	Line intercept	Modern: Jeffrey pine, mixed conifer forests, Baja California	Stephens and Gill 2005
Sierra Juarez	35.8	7.6	Ocular estimate	Modern: Jeffrey pine, Baja California	Safford, in prep

* The first modeled canopy cover value reduces crown cover by removing canopy overlap under the assumption of random distribution of stems (Crookston and Stage 1999). The value in parentheses is the total cover of all trees in the sampled plot, assuming no trees grow beneath others. This might be taken as a rough estimate of the mean maximum canopy cover in the analyzed data set.

** Values refer to 1889 and 1929 reconstructions.

[†] FVS modeling carried out post facto on size class distribution data provided in the cited paper

Table 5. Sizes of canopy gaps/regeneration patches/tree clumps in historical and contemporary reference yellow pine and mixed conifer stands in the North American Mediterranean zone and the neighboring southwestern US.

Site	Minimum size (ha)	Maximum size (ha)	Observations	Source
Kings Canyon National Park	0.014	0.16	Sequoia-mixed conifer; tree clumps	Bonnicksen and Stone 1981
Sequoia National Park	0.16	1.17	Sequoia-mixed conifer; canopy gaps	Demetry 1995
Lassen National Forest	0.003	0.14	Ponderosa pine-black oak; tree clumps	Taylor 2010
Lassen National Forest	?	0.18	Ponderosa pine; tree clumps	Youngblood et al. 2004
Lake Tahoe Basin	0.003	0.07	Jeffrey pine-white fir; tree clumps	Taylor 2004
Stanislaus National Forest	0.03	0.21	Mixed conifer; canopy gaps	Knapp 2012
Sierra de San Pedro Mártir, northern Baja California	0.001	0.07	Jeffrey pine-mixed conifer; regeneration patches	Stephens and Fry 2005
eastern Arizona	0.06	0.26	Ponderosa pine; regeneration patches	Cooper 1960
northern Arizona	0.02	0.29	Ponderosa pine; tree clumps	White 1985
central Oregon	?	0.16	Ponderosa pine; tree clumps	Youngblood et al. 2004
eastern Oregon	0.025	0.35	Ponderosa pine; tree clumps	Morrow 1985

Table 6. Estimates of percent area of six assessment area National Forests covered by seral shrub fields in 1924 and the early to mid-2000s.

National Forest	Percent in seral shrub fields	
	1924*	2000s ¹
Lassen	11.8	8.5
Plumas	11.7	6.2
Tahoe	8.6	10.6
Eldorado	7.0	9.2
Stanislaus	16.2	9.9
Sierra	11.1	7.0
Average	11.1	8.6

* Data from Table 19 in Show and Kotok (1924)

¹ Data from most recent Forest Service existing vegetation maps, using CALVEG types mixed chaparral and montane chaparral where growing in areas identified as having sufficient productivity to support forest. Imagery is from the early and mid-2000s. Data courtesy of Jay Miller, USFS.

Table 7. Shrub cover (%) in reference yellow pine and mixed conifer stands in the North American Mediterranean zone.

Site	Mean	Median	Mode	St. Dev.	Observations	Source
Northeastern assessment area	21.4	8	8	6.6	Yellow pine forests, mature and old-growth, uncut	Smith 1994
Northern assessment area	20.8	11.5	10	20.3	Mixed conifer forests, mature and old-growth, uncut	Fites 1993
Assessment area (mostly southern and central)	16.9			23	YPMC forest, uncut, with frequent and/or recent non stand-replacing fire	Lydersen and North 2012
Sierra Juarez, northern Baja California	16.3	17.5	0	13.9	Yellow pine forests, uncut, only recent fire suppression	Safford, in prep.
Sierra de San Pedro Mártir, northern Baja California	17.8	5	0	25.8	Mixed conifer forests, uncut, only recent fire suppression	Safford, in prep.

Table 8. Reference fire return intervals (FRIs; all values in years) from the centuries preceding Euroamerican settlement for yellow pine and mixed conifer forests in California, derived from literature review. Mean minimum and mean maximum are rounded to the nearest multiple of 5. Source: Van de Water and Safford (2011)

Forest type	Mean	Median	Mean Min	Mean Max	Number of sources
Yellow pine	11	7	5	40	24
Dry mixed conifer	11	9	5	50	37
Moist mixed conifer	16	12	5	80	53

Table 9. Reference fire rotations (in years) from the centuries preceeding Euroamerican settlement for yellow pine and mixed conifer forests in California. Source: Mallek et al. (in review)

Forest type	Mean	Low	High	Number of sources
Yellow pine	22	11	34	9
Dry mixed conifer	23	11	34	8
Moist mixed conifer	31	15	70	12

Table 10. Major insects and diseases found in tree species of assessment area YPMC forests.*

Vegetation Type: yellow pine and mixed conifer								
ponderosa pine, <i>Pinus ponderosa</i> (Pp); Jeffrey pine, <i>P. jeffreyi</i> (Pj); sugar pine, <i>P. lambertiana</i> (Pl); white fir, <i>Abies concolor</i> (Ac); incense cedar, <i>Libocedrus decurrens</i> (Ld); Douglas-fir, <i>Pseudotsuga menziesii</i> (Pm); California black oak, <i>Quercus kelloggii</i> (Qk); giant sequoia, <i>Sequoiadendron giganteum</i> (Sg)								
Agent	Host							
	Pp	Pj	Pl	Ac	Ld	Pm	Qk	Sg
Heterobasidion root disease, <i>Heterobasidion</i> spp.	x	x	x	x	x	x		x
black stain root disease, <i>Leptographium wageneri</i>	x	x	x			x		
Armillaria root disease, <i>Armillaria</i> spp.	x	x	x	x	x	x	x	x
dwarf mistletoe, <i>Arceuthobium</i> spp.	x	x	x	x		x		
white pine blister rust, <i>Cronartium ribicola</i>			x					
western pine beetle, <i>Dendroctonus brevicomis</i>	x							
Jeffrey pine beetle, <i>Dendroctonus jeffreyi</i>		x						
mountain pine beetle, <i>Dendroctonus ponderosae</i>	x		x					
fir engraver beetle, <i>Scolytus ventralis</i>				x				
Douglas-fir beetle <i>Dendroctonus pseudotsugae</i>						x		
Douglas-fir tussock moth, <i>Orgyia pseudotsugata</i>				x				
flatheaded fir borer, <i>Melanophila drummondi</i>				x		x		

* Table courtesy of Sherri Smith, Regional Entomologist

Draft Summary of probable deviations from NRV. Yellow pine and mixed conifer forests. Date: 5-16-2013

Ecosystem attribute	Indicator group	Indicator	Variable	Within NRV?*	Confidence	Discussion. See text on pp.:	Notes
Composition	Functional diversity	Functional groups/ growth forms	Proportion of shade tolerant vs. shade intolerant spp.	No	High	TBD	Major shift from dominance of shade intolerant species to dominance of shade tolerant species
Composition	Species diversity	Species richness	Plant species richness	Yes	Medium	TBD	No evidence of major species loss
Function	Disturbance	Fire	Fire regime	No	High	TBD	Shift from Fire Regime I to Fire Regimes III and IV
Function	Disturbance	Fire	Fire frequency	No	High	TBD	Current frequency far below presettlement but rising
Function	Disturbance	Fire	Fire severity	No	Medium to High	TBD	Current severity higher than presettlement and rising
Function	Disturbance	Fire	Fire size	No	Medium	TBD	Current mean and mean max fire sizes larger than presettlement mean
Function	Disturbance	Fire	High severity patch size	No	Medium	TBD	Current high severity patch sizes higher than presettlement mean and rising
Function	Disturbance	Fire	Fire rotation	No	High	TBD	Fire rotations much longer today than presettlement

Ecosystem attribute	Indicator group	Indicator	Variable	Within NRV?*	Confidence	Discussion. See text on pp.:	Notes
Function	Disturbance	Fire	Fire season	No	Medium	TBD	Fire season is becoming longer but general seasonal patterns are similar
Function	Disturbance	Fire	Annual area burned	No	High	TBD	Current mean annual area burned is much lower than all estimates of presettlement area
Function	Disturbance	Insect outbreaks	Tree mortality due to insects	unknown		TBD	Little in the way of presettlement estimates. Some conclusions can be drawn from comparisons of range of current conditions
Structure	Physiognomy	Canopy cover	Percent cover	No	Medium	TBD	Modern mean canopy cover is above presettlement
Structure	Physiognomy	Coarse woody debris (CWD)	Pieces of CWD per unit area	No	Medium	TBD	Density of CWD is higher in contemporary forests
Structure	Physiognomy	Coarse woody debris (CWD)	Mass of CWD per unit area	unknown		TBD	Yet to be assessed
Structure	Physiognomy	Forest fuels	Tons/ha	unknown		TBD	Yet to be assessed
Structure	Physiognomy	Functional groups/ growth forms	Proportion early/ middle/late seral forest	No	Medium	TBD	Current lack of old forest successional stages, perhaps some localized lack of early stages

Ecosystem attribute	Indicator group	Indicator	Variable	Within NRV?*	Confidence	Discussion. See text on pp.:	Notes
Structure	Physiognomy	Gap size distribution	Gap size	No	Medium	TBD	Gap sizes are generally decreasing (in undisturbed forests), but also increasing in disturbed forests due to more severe disturbance
Structure	Physiognomy	Grass and forb cover	% cover	Maybe	Low	TBD	Difficult to assess, little presettlement data. Overall herbaceous cover on landscape may be similar; cover within forest stands may be lower due to fire suppression
Structure	Physiognomy	Overstory density	Number of trees per unit area	No	High	TBD	Current density higher on average than presettlement
Structure	Physiognomy	Overstory density	Number of large trees per unit area	No	High	TBD	Large tree density is lower in modern forests
Structure	Physiognomy	Shrub cover	% cover	Maybe	Low	TBD	Difficult to assess, little presettlement data. Overall shrub cover on landscape not much changed over time; cover within forest stands may be lower due to fire suppression
Structure	Physiognomy	Snag density	Number of snags per unit area	No	Medium	TBD	Snag density is higher in contemporary forests

Ecosystem attribute	Indicator group	Indicator	Variable	Within NRV?*	Confidence	Discussion. See text on pp.:	Notes
Structure	Physiognomy	Tree size class distribution	Tree size class distribution	No	High	TBD	Major increases in small size classes, and general decreases in large size classes. Change in distribution shape from +/- flat or hump-shaped, to strongly J-shaped.
Structure	Productivity	Tree basal area	Basal area	Yes	Medium	TBD	Basal area similar or higher in modern forests.

1 = NRV presettlement reference period is assumed to refer to 1500/1600 to 1850, unless otherwise indicated in Notes. NRV for most indicators/variables also includes information from contemporary reference sites.

* = as defined as the range of means from multiple sources

FIGURES

Figure 1. Distribution of YPMC forests in the assessment area, with climate station locations indicated.

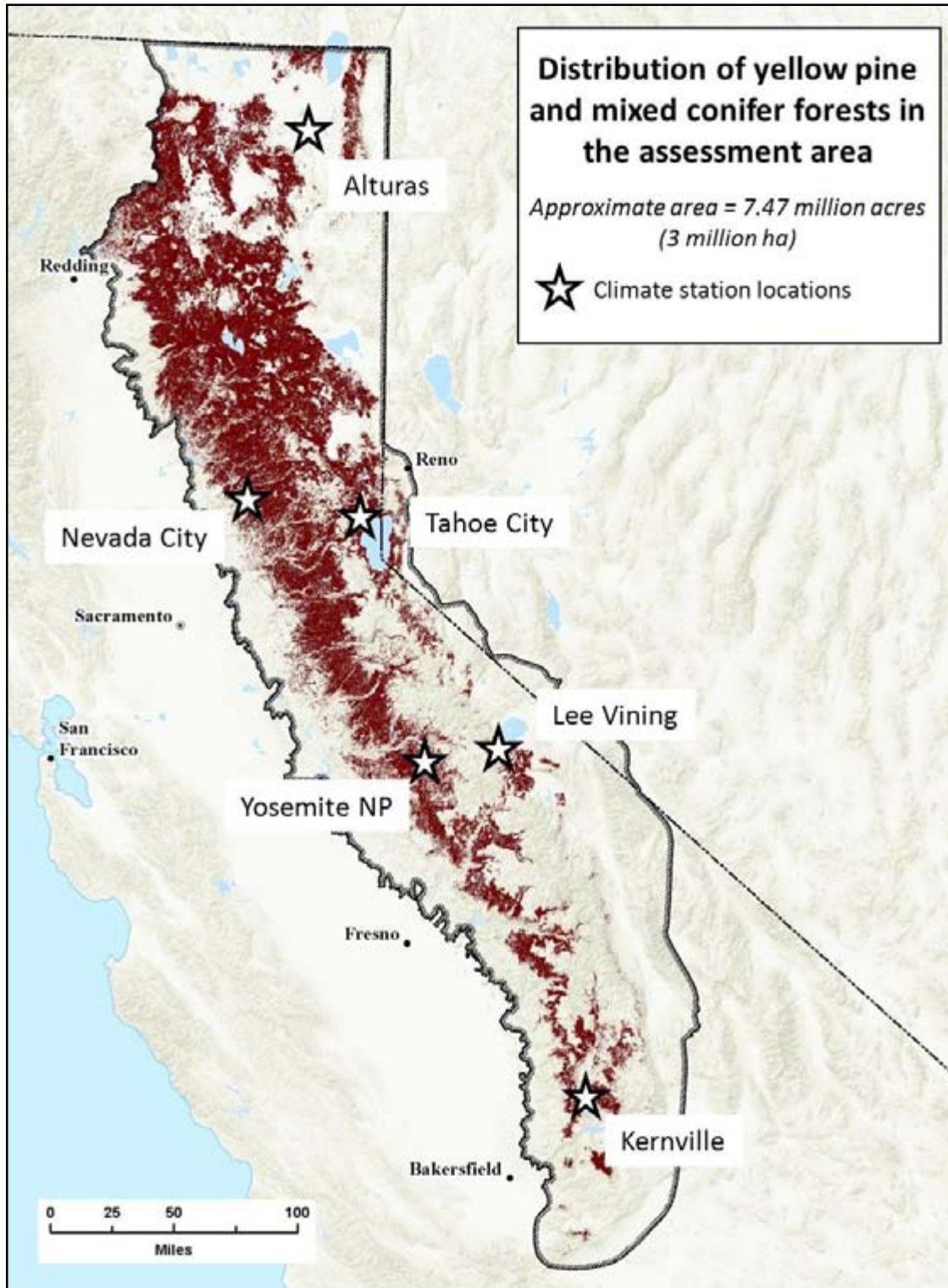


Figure 2. Walther type climate diagrams for six stations in or near YPMC forest in the assessment area; elevations provided. Locations shown in Figure 0. Dark line represents precipitation (left Y-axis, mm), Dotted line is mean monthly temperature (right Y-axis, °C). Dry season length is approximately the period during which the precipitation curve undercuts the temperature curve.

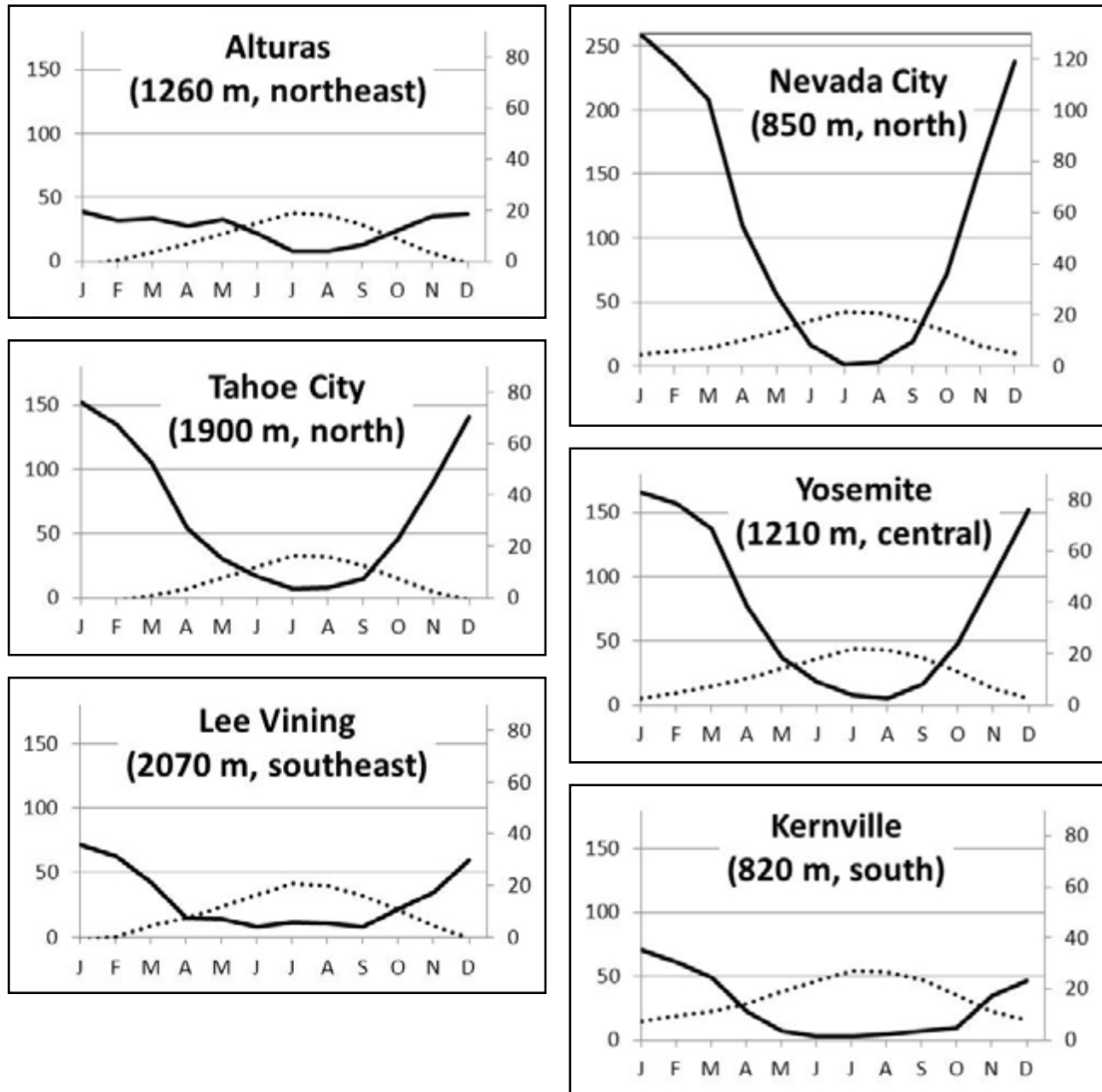


Figure 3. Comparison of predicted small tree (dbh between 3.8 and 7.6 cm) growth rates of yellow pines, sugar pine, and true firs as a function of the basal area of trees larger than the subject tree. Curves shown are for a site of moderate productivity (site index of 80) and crown ratio of 50%.

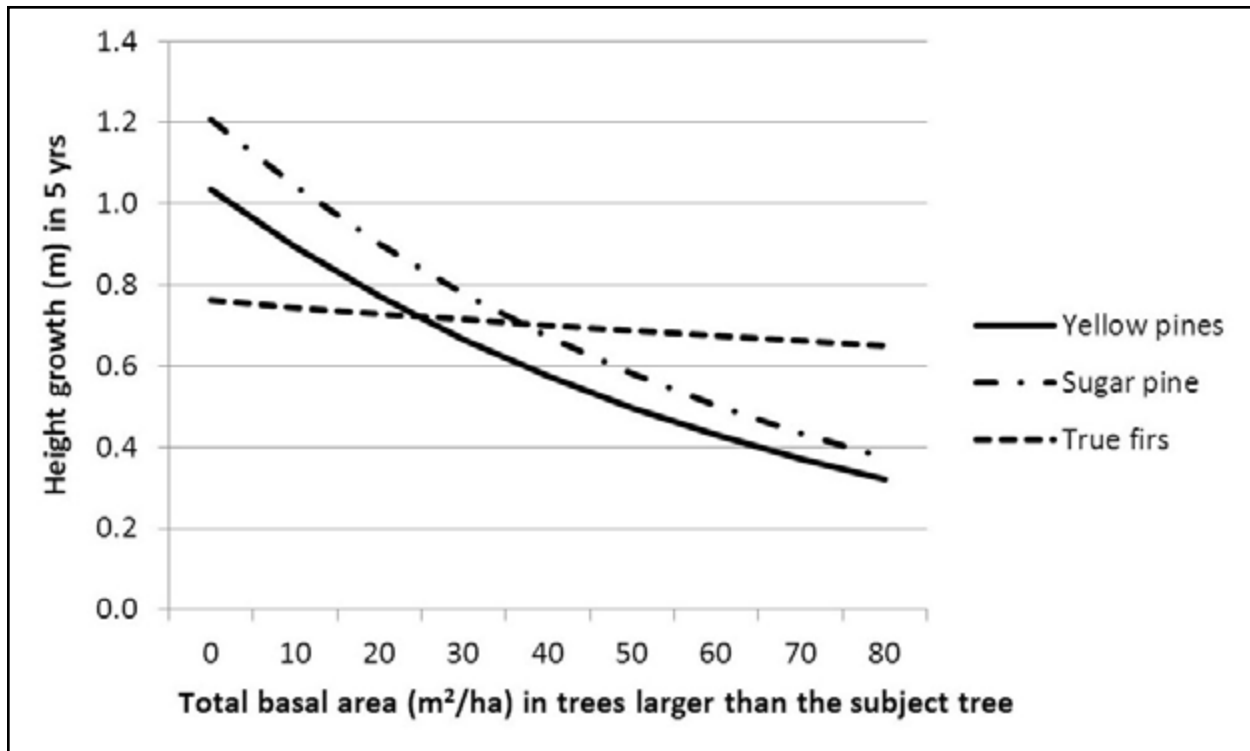


Figure 4. Bark thickness of six tree species in the assessment area, as a function of diameter at breast height (DBH). Thickness calculated based on formulas in Dolph 1984. ABCO = *Abies concolor* (white fir); CADE = *Calocedrus decurrens* (incense cedar); PIJE = *Pinus jeffreyi* (Jeffrey pine); PILA = *P. lambertiana* (sugar pine); PIPO = *P. ponderosa* (Ponderosa pine); PSME = *Pseudotsuga menziesii* (Douglas-fir).

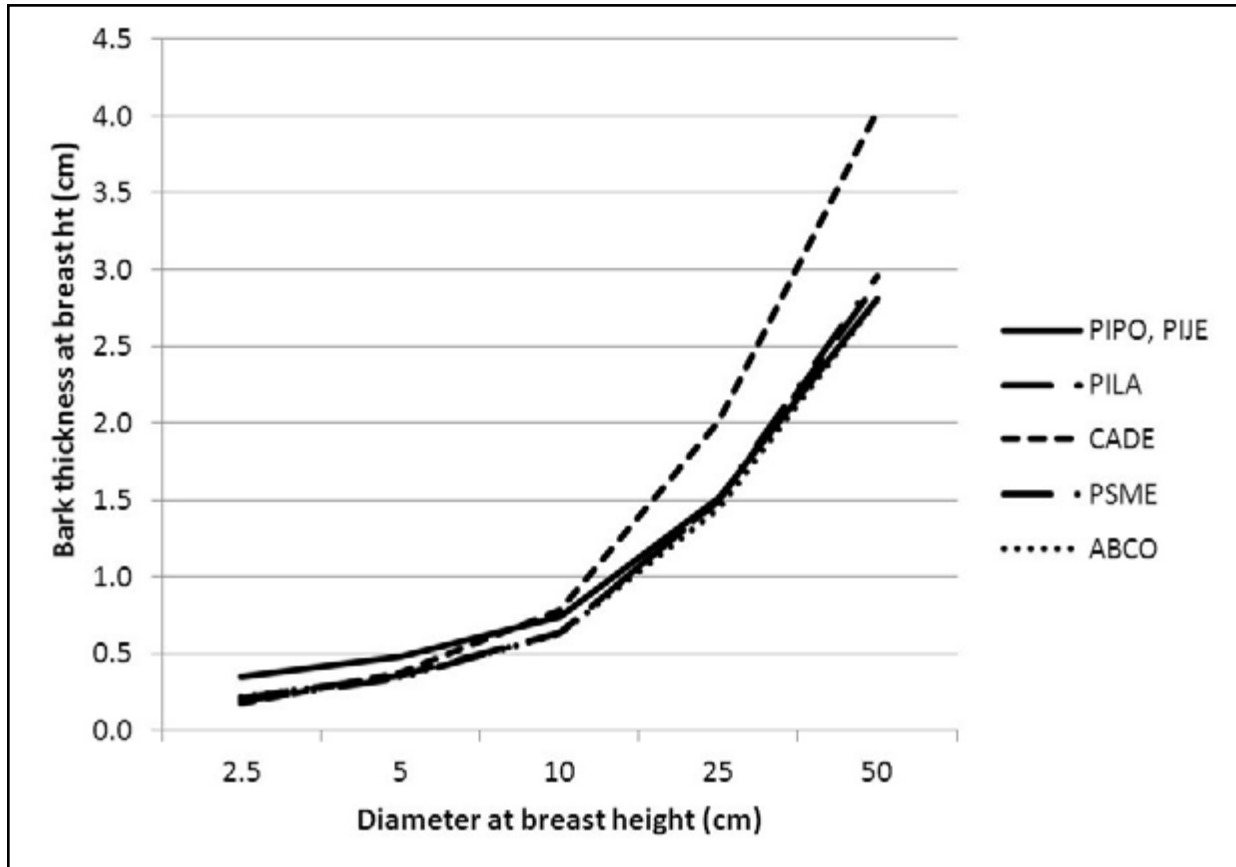


Figure 5. Mean percent fire return interval departure, generalized to the ecological subregion (Miles and Goudey 1997) from Forest Service and Park Service lands found within each subregion. Warm colors are experiencing more fire than under pre-Euroamerican conditions, cool colors are experiencing less fire. Figure 11 from Safford and Van de Water (in press).

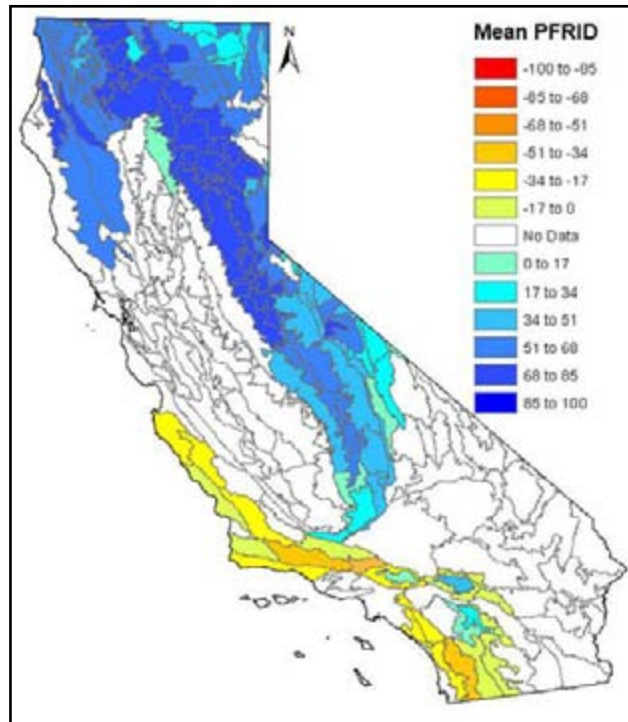


Figure 6. Comparison of average fire severity proportions measured in fires 1984-2004 in the assessment area vs. reference (presettlement) estimates for four major forest types and a composite for the northern half of the assessment area. Reference/presettlement estimates from LANDFIRE BpS modeling (Rollins and Frame 2006, Rollins 2009), except for northern Sierra, which are empirical estimates from Leiberg (1902; see text). Forest types organized from driest and lowest on the left to highest and wettest on the right. Eastside pine and ponderosa pine are both types of yellow pine forest; white fir belongs primarily within moist mixed conifer. Red fir pictured to show how higher elevation forests are not strongly departed from presettlement severity patterns. Figure 11 from Miller and Safford (2008).

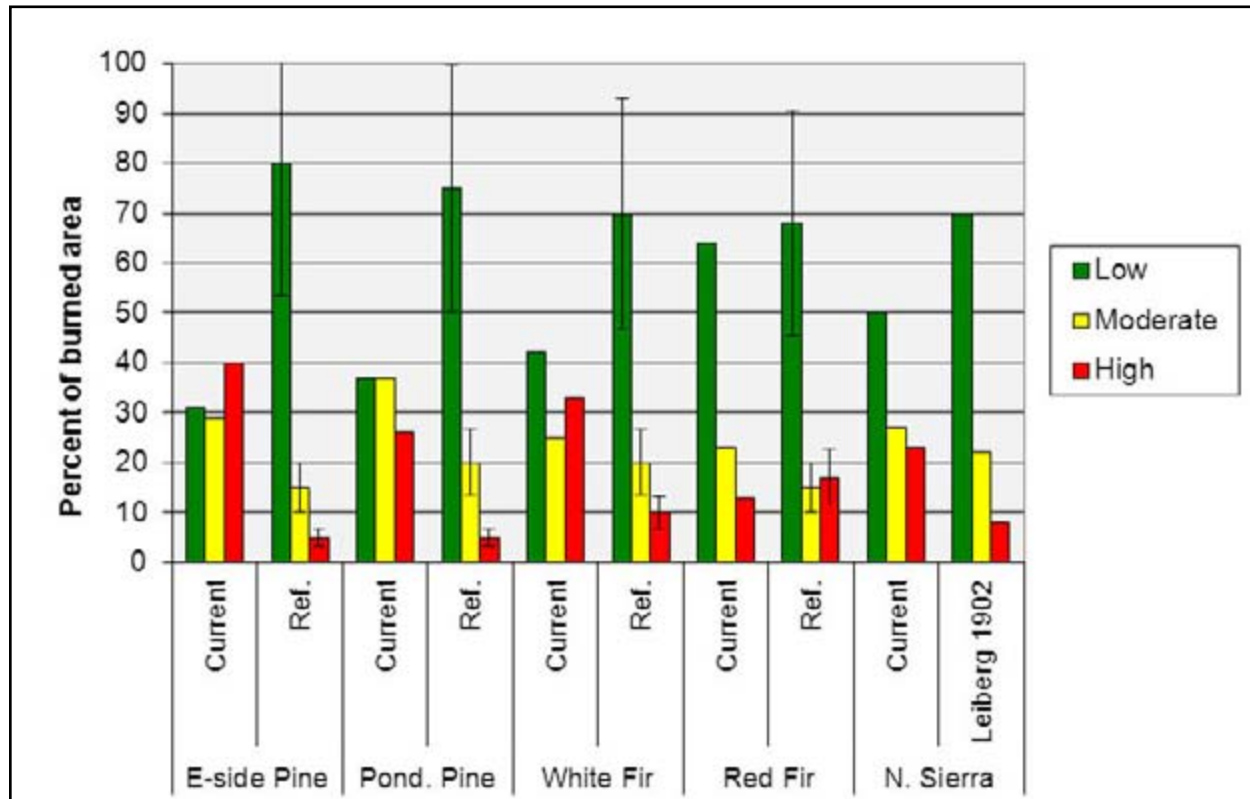


Figure 7. Temporal trends in percent high severity fire for YPMC forests in the assessment area between 1984 and 2010. The figure shows annual percentages of high severity fire, the time series (ARIMA) model for the 1984-2010 period, and linear trend lines for seven time series beginning in 1984 but ending in 2004 to 2010. Figure 11 from Miller and Safford (2012).

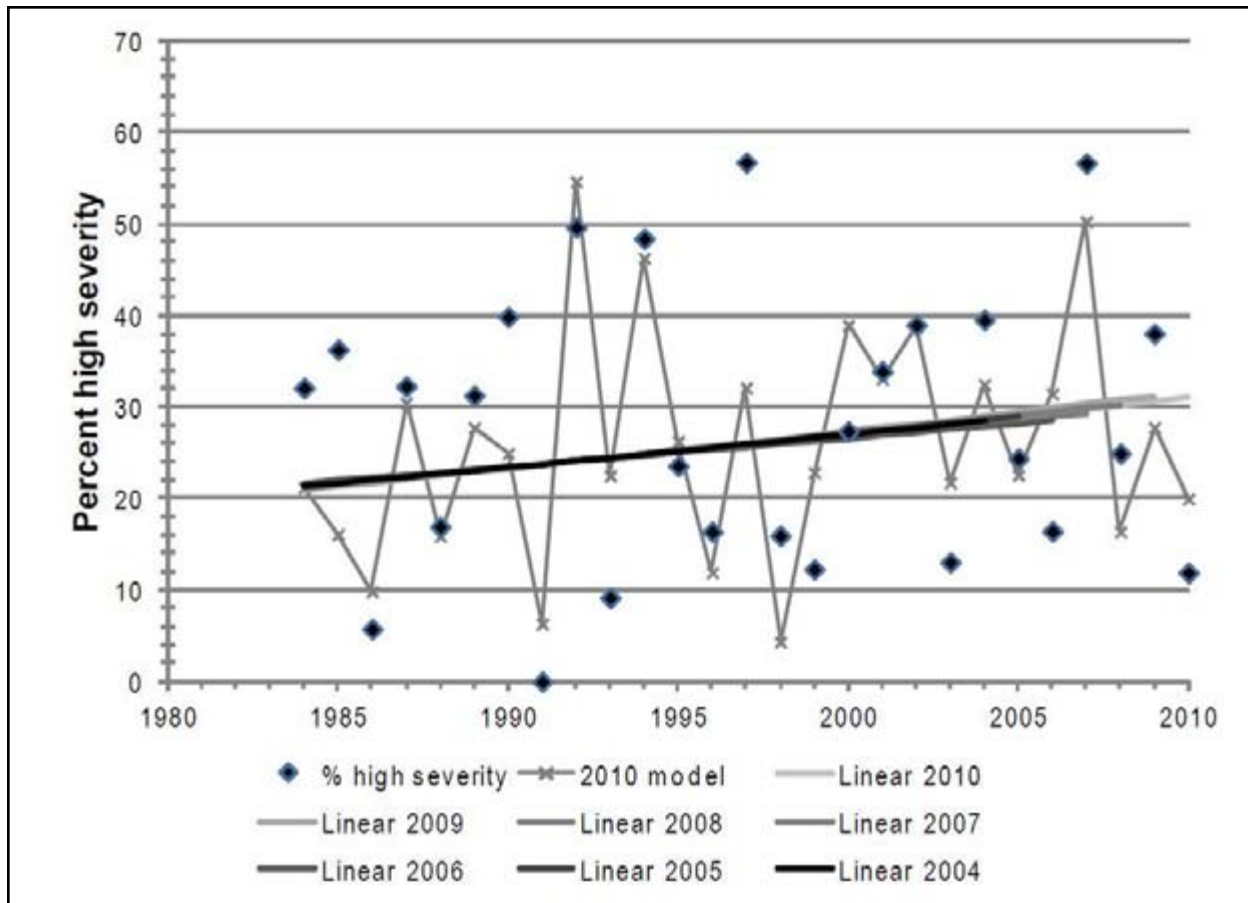


Figure 8. Comparison of mean fire sizes from YPMC reference sites with intact fire regimes and the assessment area (forest fires only) since the institution of fire suppression in the early 20th century. Only fires ≥ 10 ha included. Ten sources used for historical reference numbers, see text. SSPM = Sierra San Pedro Mártir, data from Minnich et al. 2000. Illilouette watershed wildland fire use fires from B. Collins, pers. comm. 20th-21st century fires from California fire perimeter database.

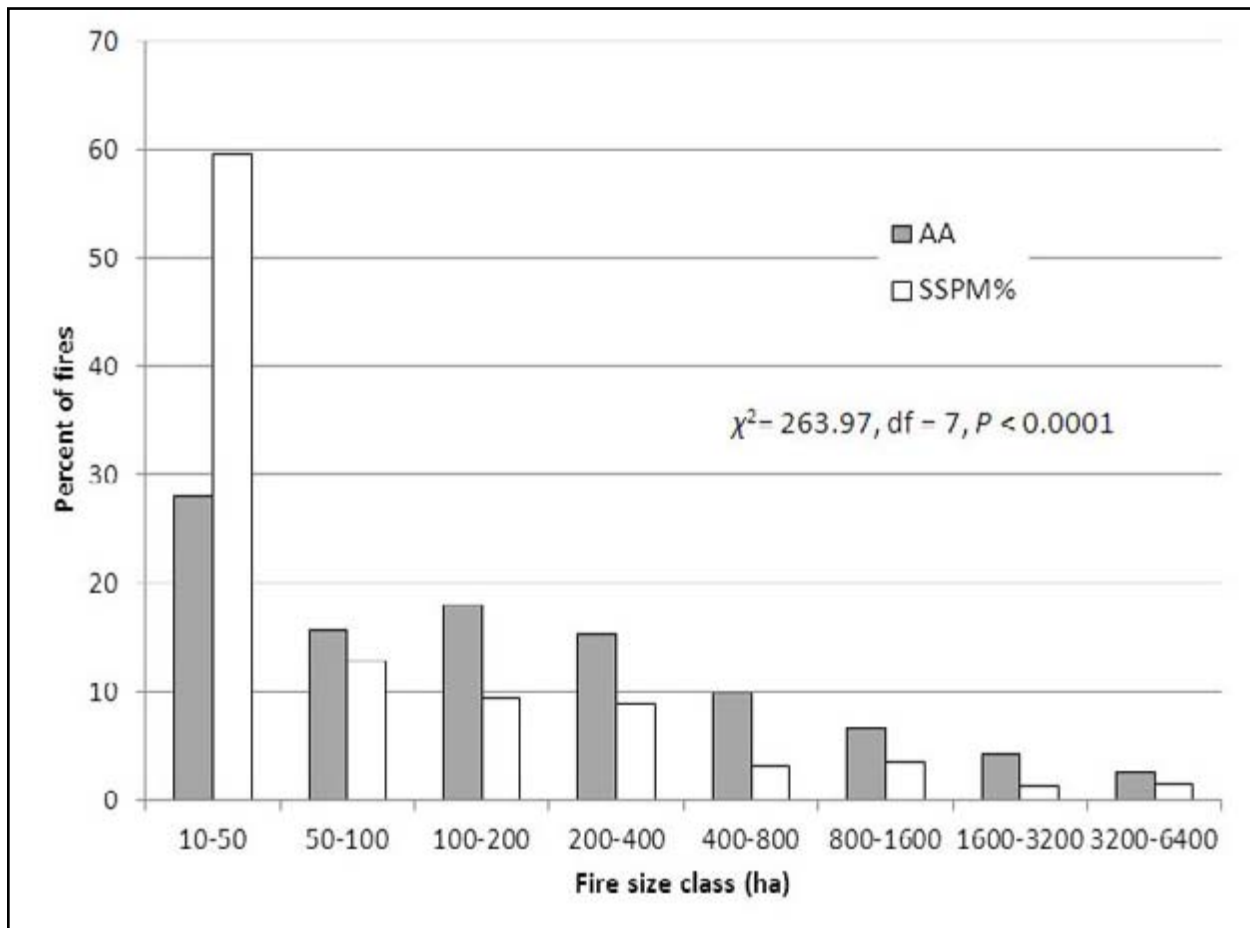


Figure 9. Comparison of 20th century fire size frequency distributions, assessment area (AA) vs. Sierra San Pedro Mártir (SSPM). Proportionally speaking, the assessment area experiences many more large fires, and many fewer small fires. Only fires >10 ha in size included.

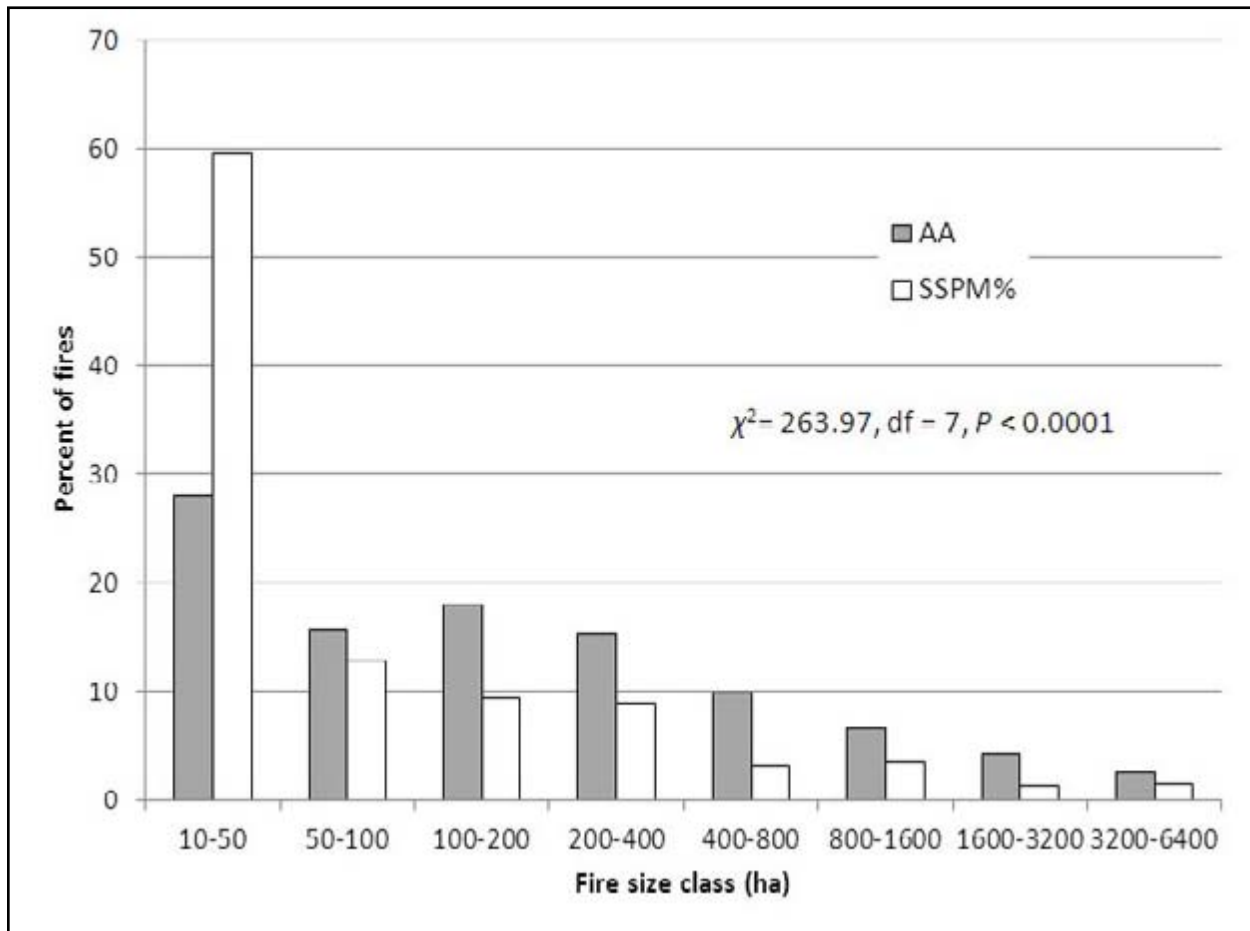


Figure 10. Lenihan et al. (2008) modeling results for the Sierra Nevada and Sierra Nevada Foothills Ecological Sections, current vs. future projections of vegetation extent. These Ecological Sections include most of the Sierra Nevada west slope. The GFDL-B1 scenario = moderately drier than today, with a moderate temperature increase (<5.5° F); PCM-A2 = similar ppt. to today, with <5.5° temp. increase; GFDL-A2 = much drier than today and much warmer (>7.2° higher). All scenarios project significant loss of subalpine and alpine vegetation. Most scenarios project lower cover of shrubland (including west side chaparral and east side sagebrush), due principally to increasing frequencies and extent of fire. Large increases in the hardwood component of forests are projected in all scenarios except for the hot-dry scenario in the Foothills. Large increases in cover of grassland are projected for the Sierra Nevada section. The drier scenarios project moderate expansion of arid lands. In the Sierra Nevada section, conifer forest decreases in cover under all scenarios. Figure from Safford et al. (2012b).

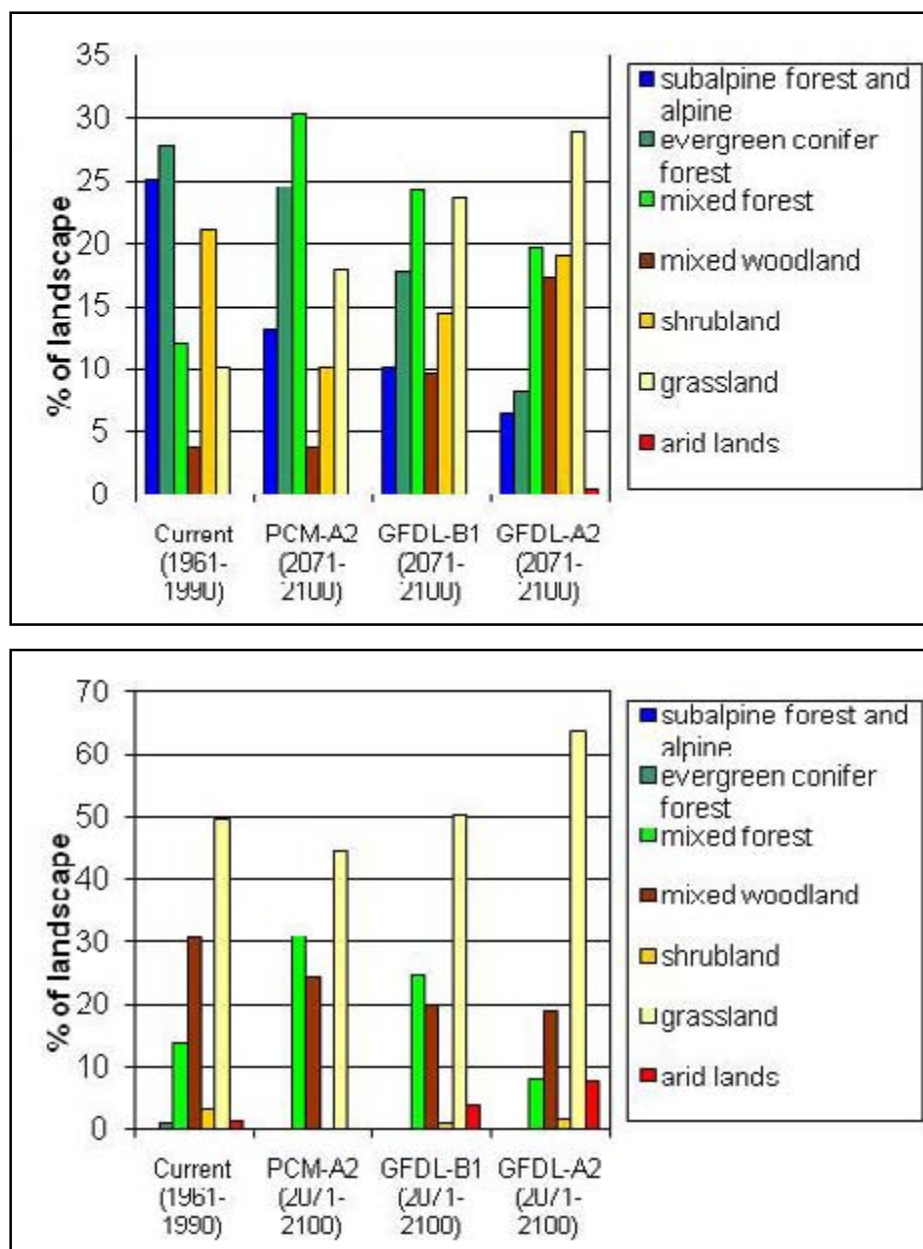


Figure 11. Box plots of upper and lower quartiles for basal area of five tree species modeled by Kercher and Axelrod (1984) for a YPMC forest at 1524 m elevation with a mean fire return interval of 7 years. Values are from temporal distributions of basal areas as predicted by the SILVA forest succession model, averaged over 10 runs of 500 years. Horizontal lines within quartile boxes represent the median, the “error bars” represent upper and lower ranges for each species. Douglas-fir was also modeled but only accounted for about 1% of basal area. Figure redrawn from Fig. 6 in Kercher and Axelrod (1984)

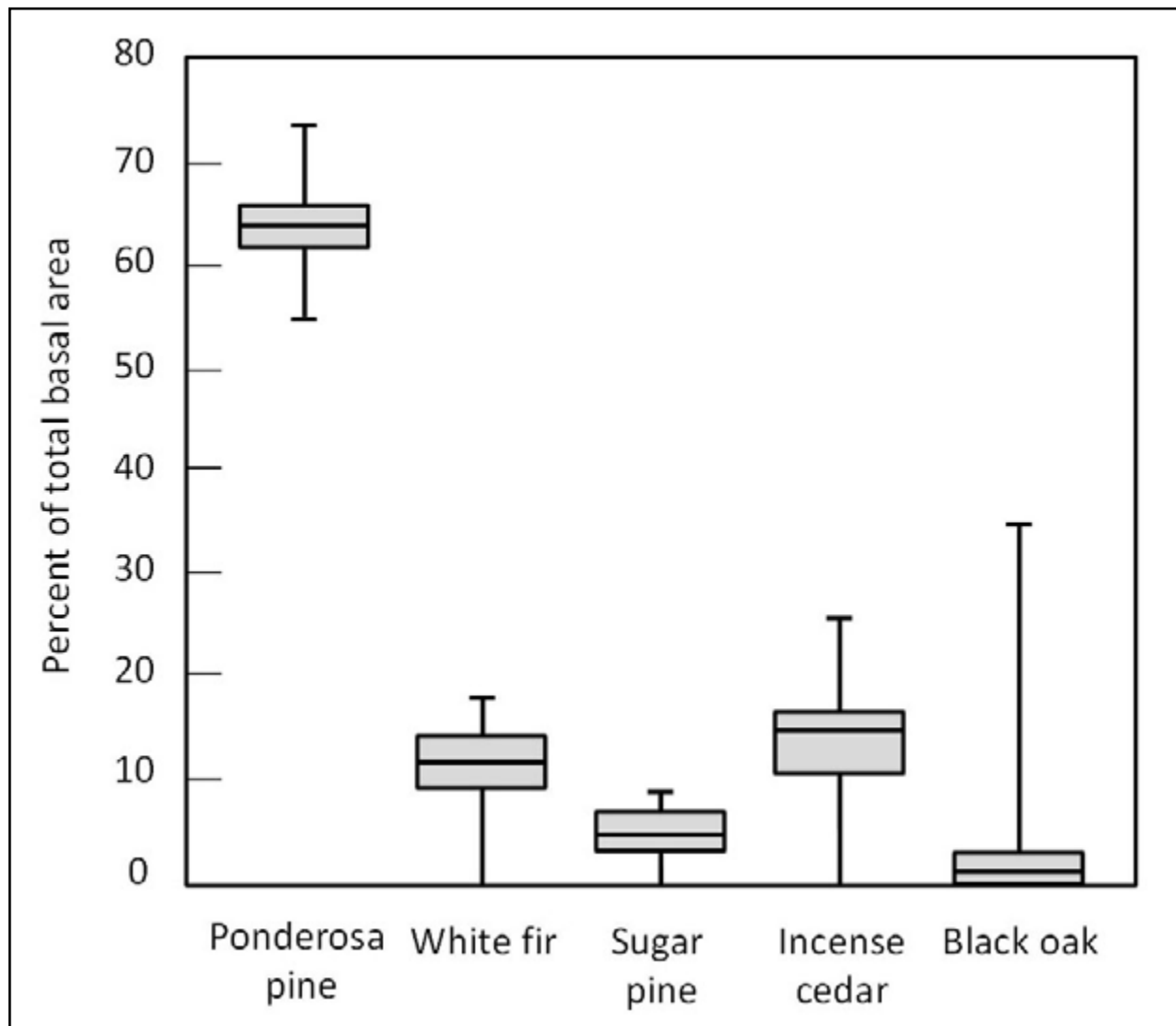


Figure 12. Composite of SILVA simulation results for five tree species in a YPMC forest at 1524 m elevation, under a mean fire return interval of 7 years. Douglas-fir was also simulated but accounted for very little of the basal area. Data from Kercher and Axelrod (1984).

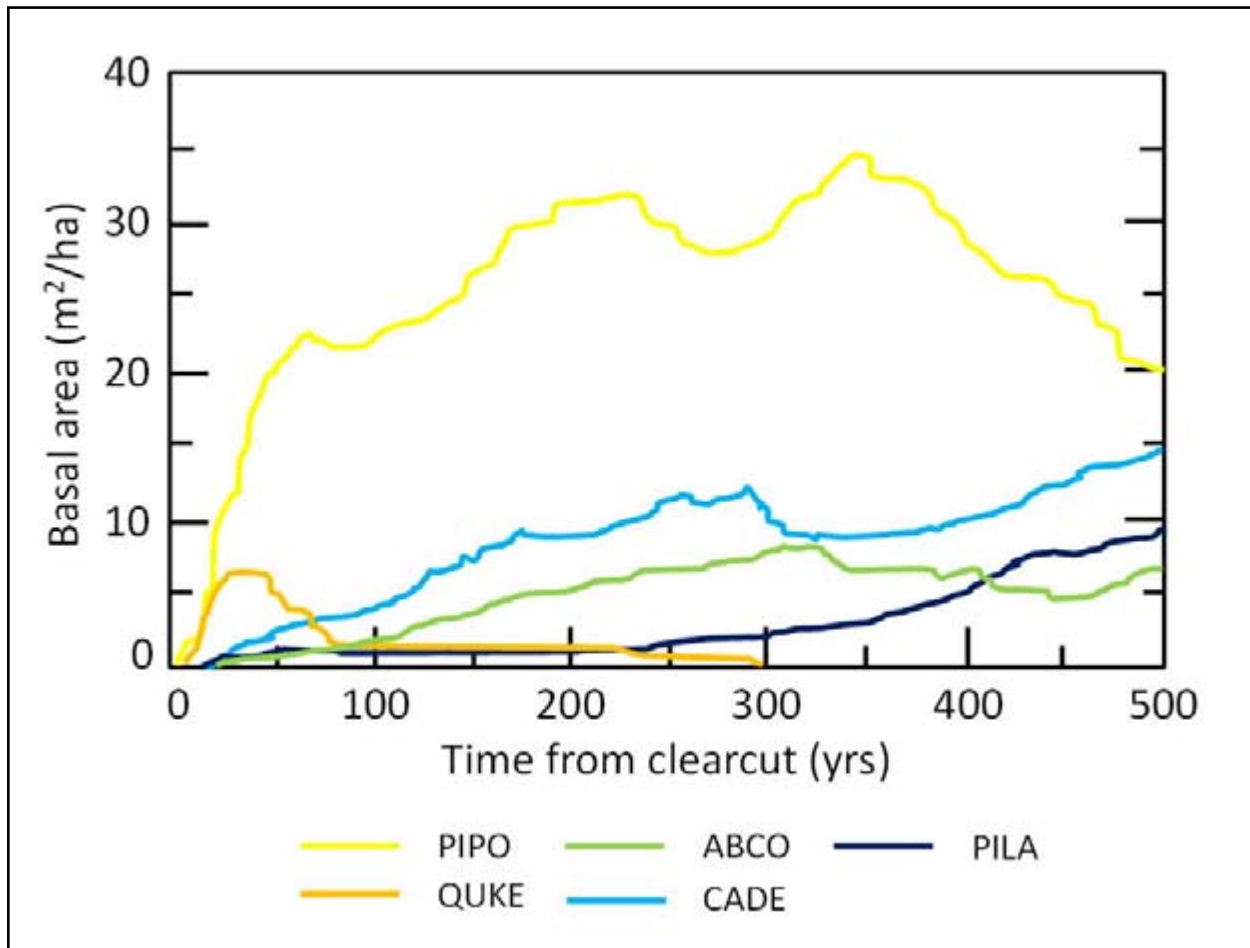


Figure 13. Average landscape conditions for presettlement YPMC forests in the assessment area, as predicted by LANDFIRE BpS state and transition models for LANDFIRE modeling region 6. Only applicable on landscapes greater than about 5000 hectares in area. See text for definitions of successional classes.

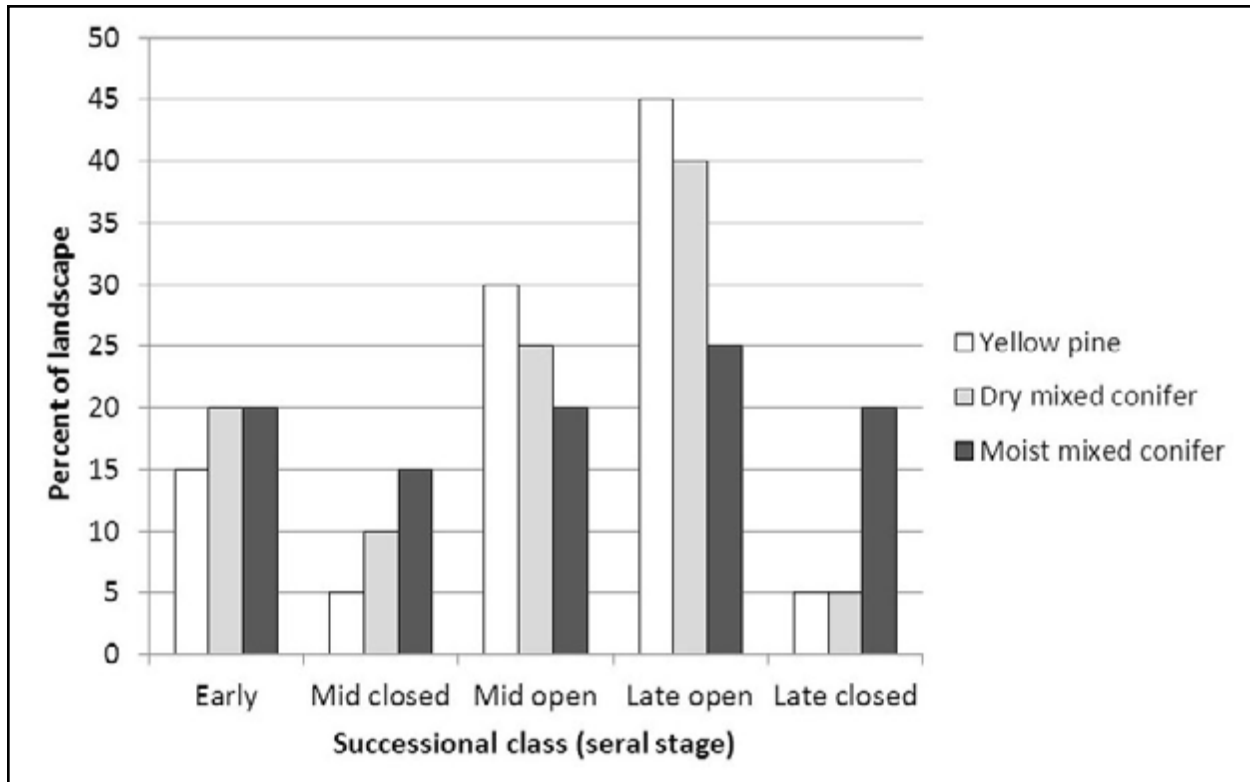


Figure 14. Comparisons of current and historical tree densities at eight YPMC sites in the assessment area. 1: Taylor 2004 (≥ 10 cm dbh); 2: North et al. 2007 (trees ≥ 5 cm); 3: Lydersen et al., in press (≥ 10 cm); 4: Scholl and Taylor 2010 (≥ 10 cm); 5: USDA 1911 vs. current from Scholl and Taylor (≥ 15.2 cm); 6: Collins et al. 2011 (≥ 15.2 cm). 7 and 8: Parsons and Debenedetti 1979 (≥ 12 cm).

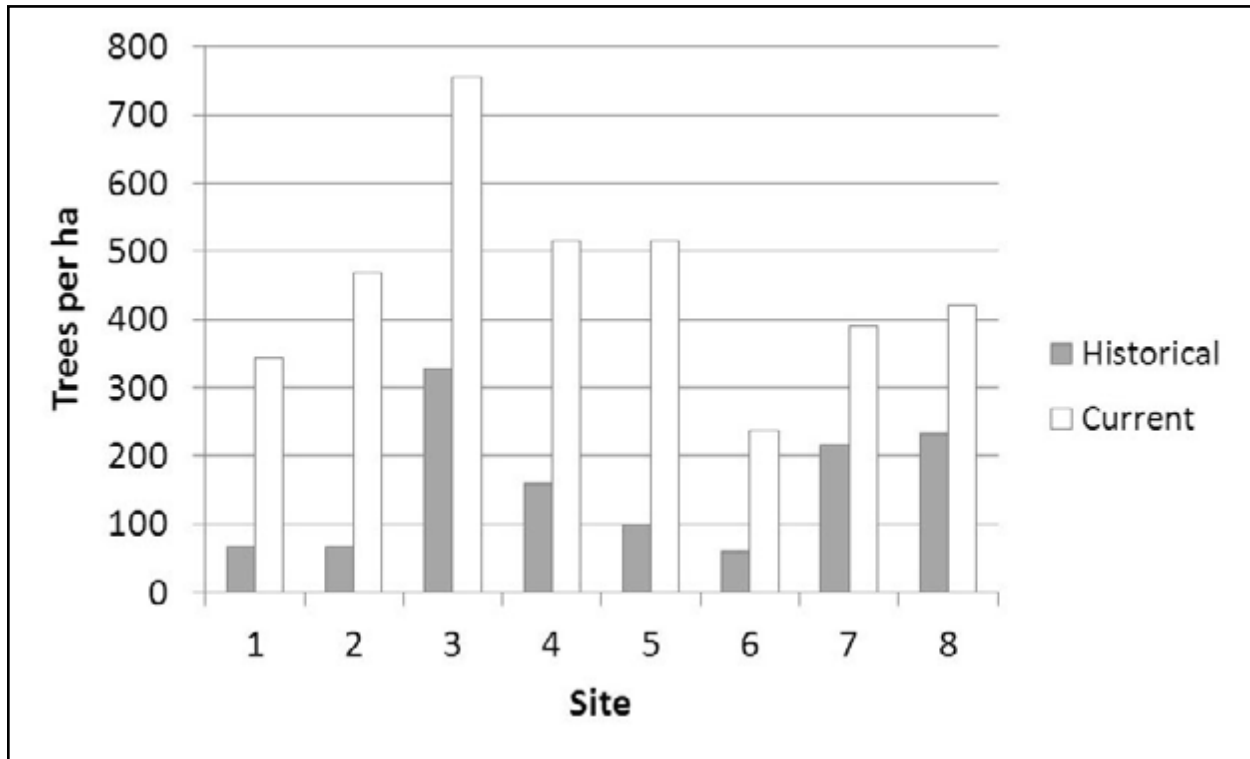


Figure 15. Change in density (trees/ha) between Vegetation Type Mapping (VTM) plots and Forest Inventory and Analysis (FIA) plots for all species combined, for seven elevation bands (Elev1: 0-499m; Elev2: 500-999m; Elev3: 1000-1499m; Elev4: 1500-1999m; Elev5: 2000-2499m; Elev6: 2500-2999m; Elev7: >3000m), and by size class: Size Class 1 (SC1; 10.2-30.4 cm dbh), Size Class 2 (SC2; 30.5-60.9 cm), Size Classes 3 and 4 (SC3&4; > 61.0 cm), and the net percent change in density from VTM to FIA plots, across all size classes. Statistically significant differences are indicated by * = $0.01 < p < 0.05$; ** = $0.001 < p \leq 0.01$; and *** = $p < 0.001$. YPMC forests are found principally in elevation bands 2-4. From Dolanc et al. (in review).

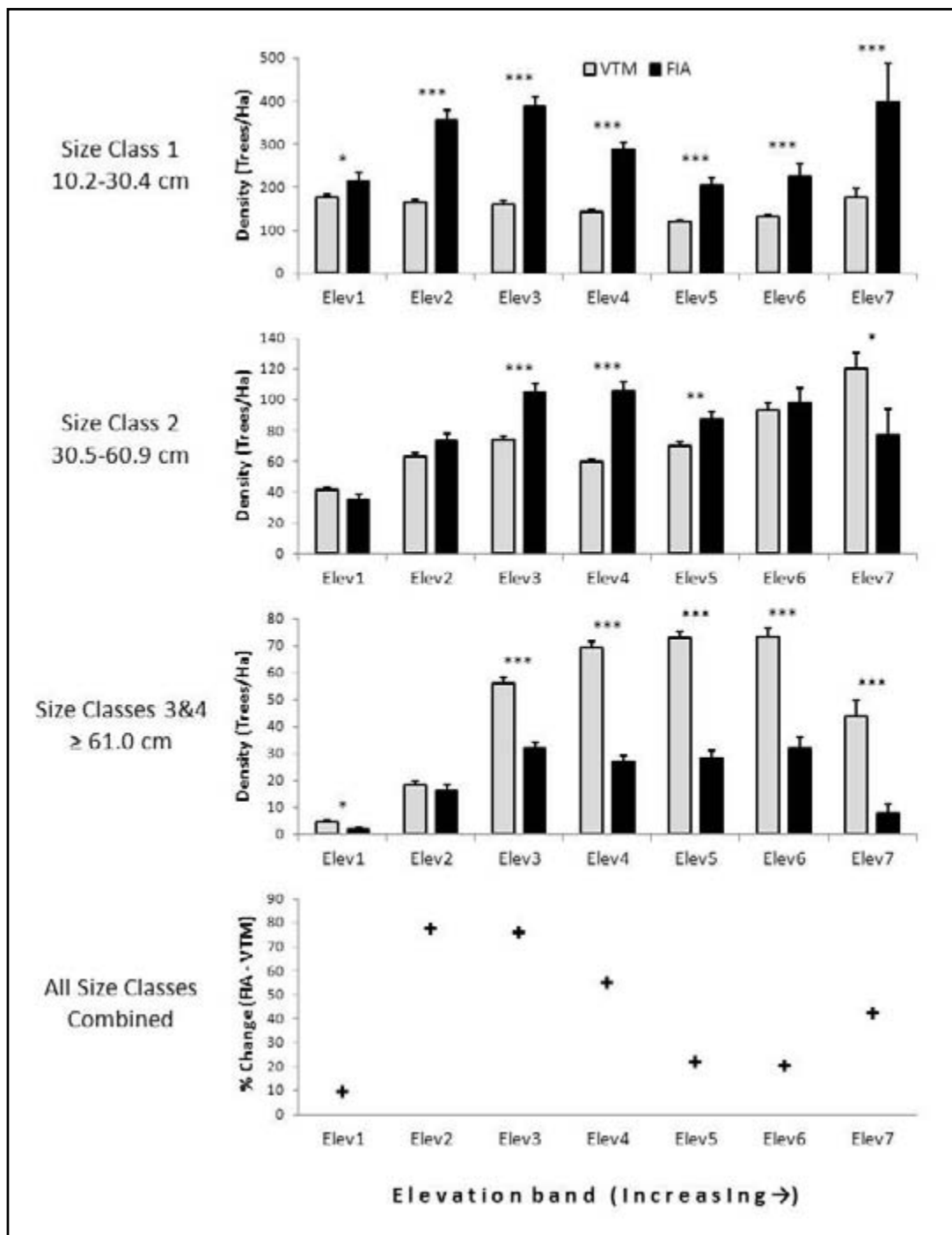


Figure 16. Historical tree size distributions from four studies of YPMC forest in the assessment area, standardized by percent of trees measured. Top – SNF: Sierra National Forest, 1865 (North et al. 2007); LTB-W: Lake Tahoe west shore, 1873 (Taylor et al., in press). Bottom – LTB-E: Lake Tahoe east shore, 1870-1900 (Taylor 2004); YNP – Yosemite National Park, 1899 (Scholl and Taylor 2010).

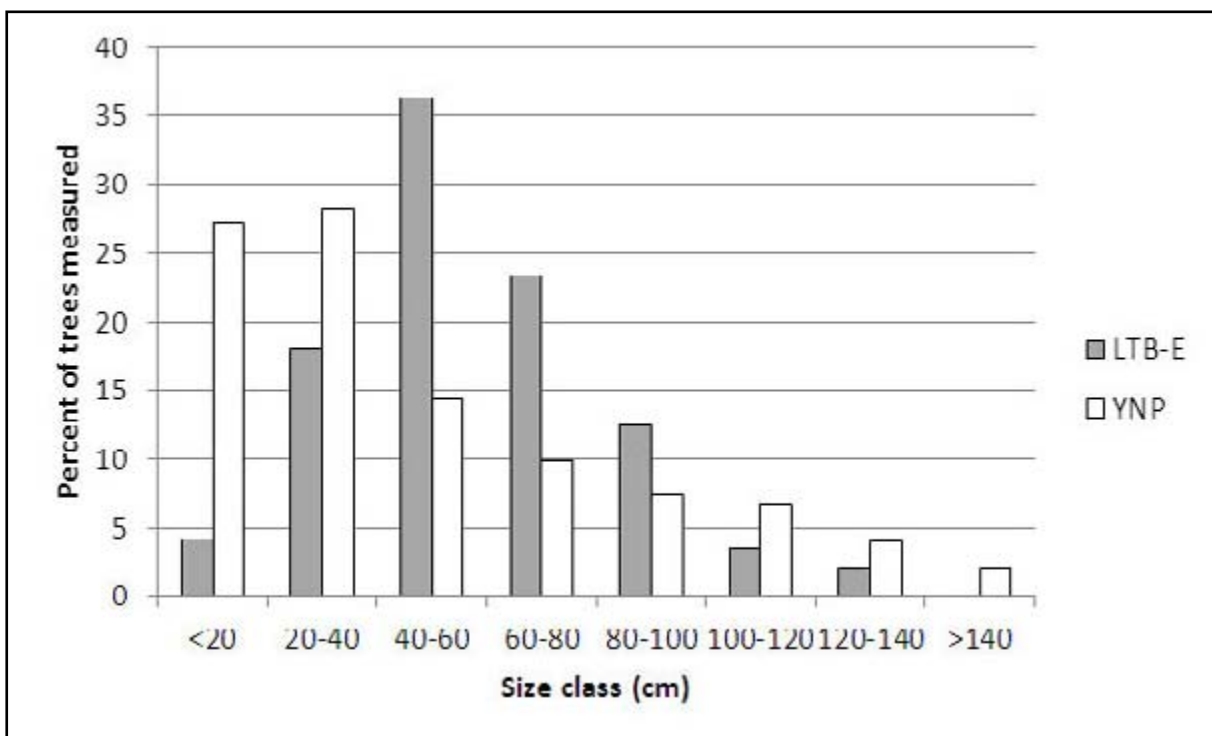
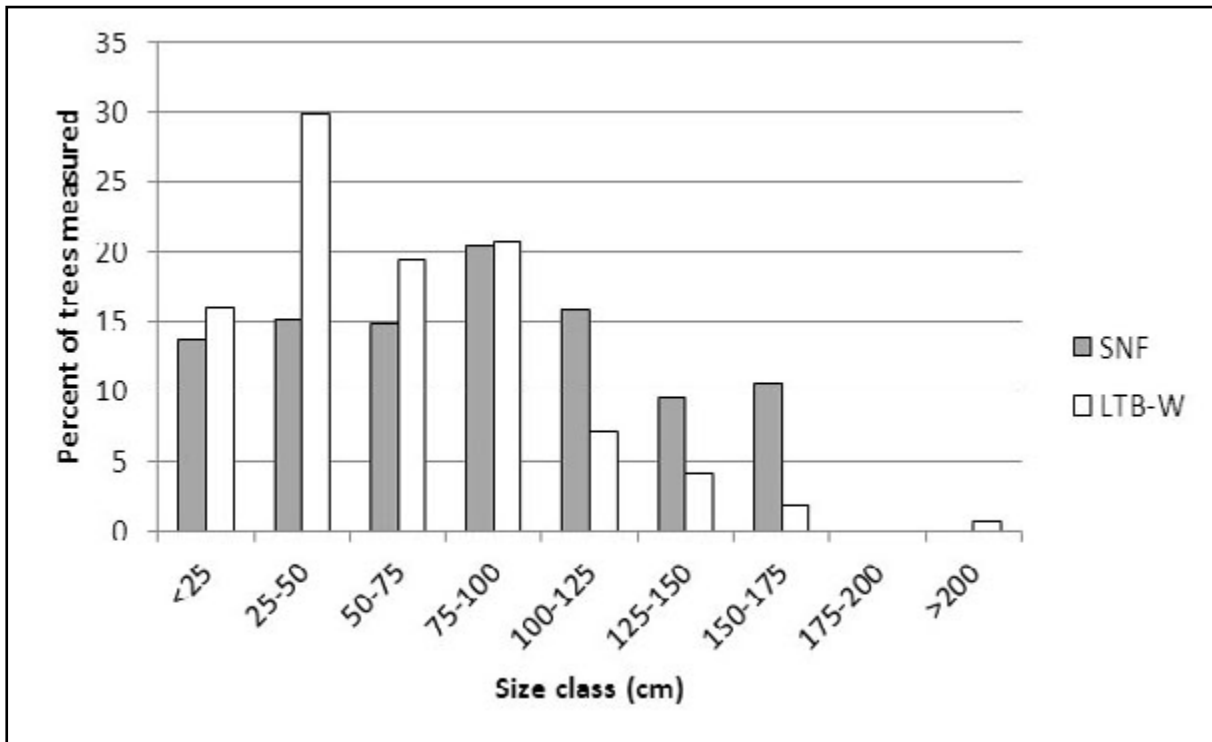


Figure 17. Modern size class distributions from three unlogged reference areas with (partially) intact fire regimes through the 20th century. “Assessment Area” = Lydersen and North 2012; LNF = Lassen National Forest (Beaver Creek Pinery), Taylor 2010; SSPM = Sierra San Pedro Mártir, Stephens and Gill 2005.

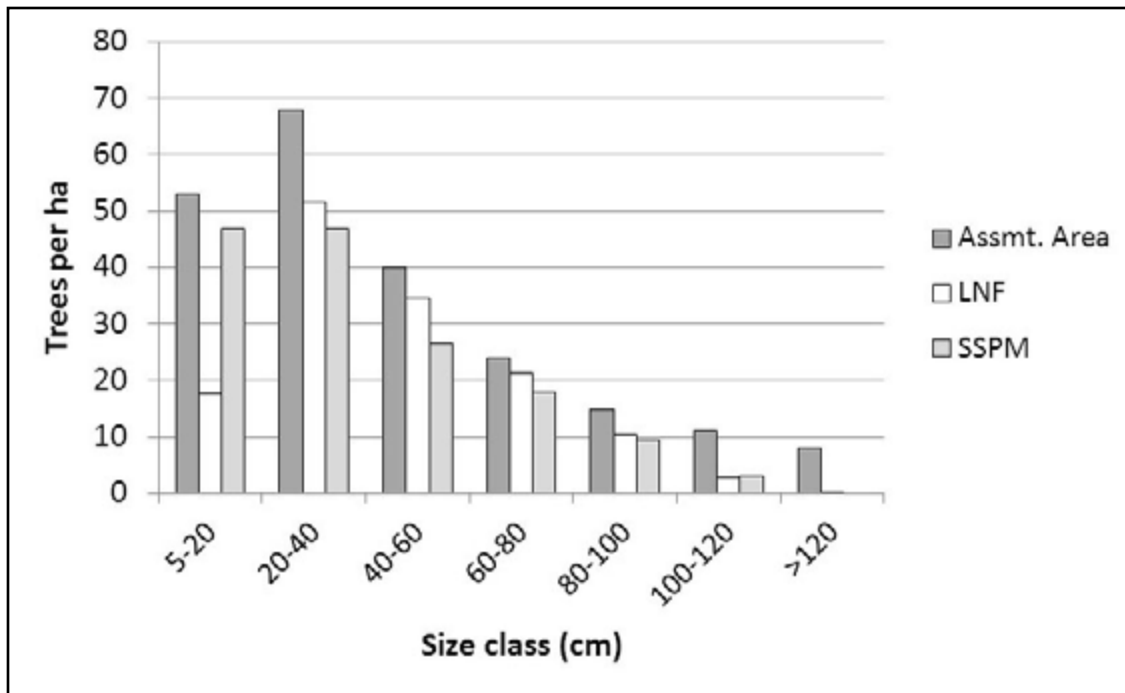


Figure 18. Comparisons of reconstructed historical tree size class distributions (blue) and contemporary distributions (red) from three sites in the assessment area. Top from North et al. 2007; middle from Taylor 2004; bottom from Scholl and Taylor 2011.

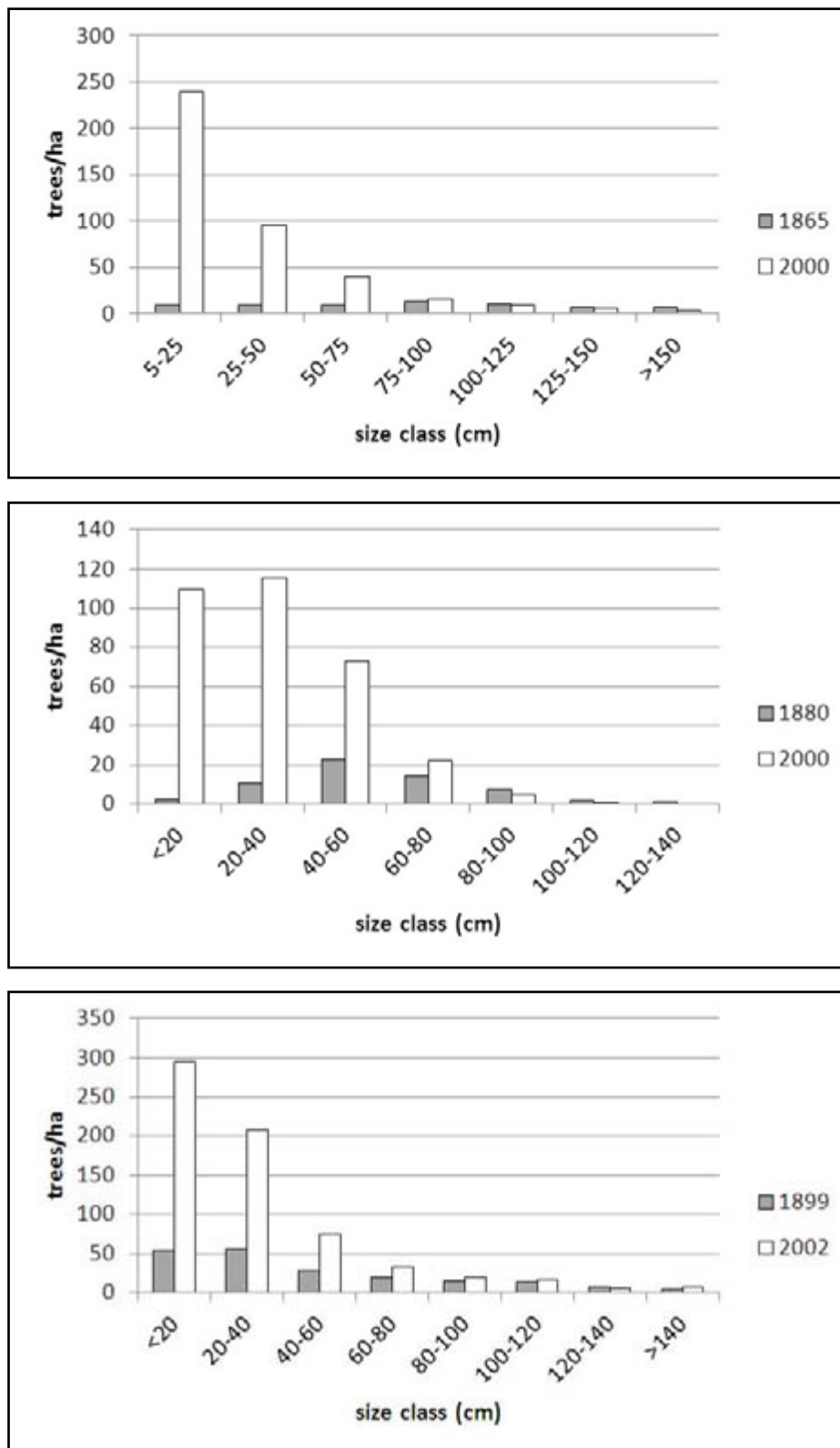


Figure 19. Average size class distribution for compilation of most recent FIA plot data in assessment area YPMC forests (USFS 2013).

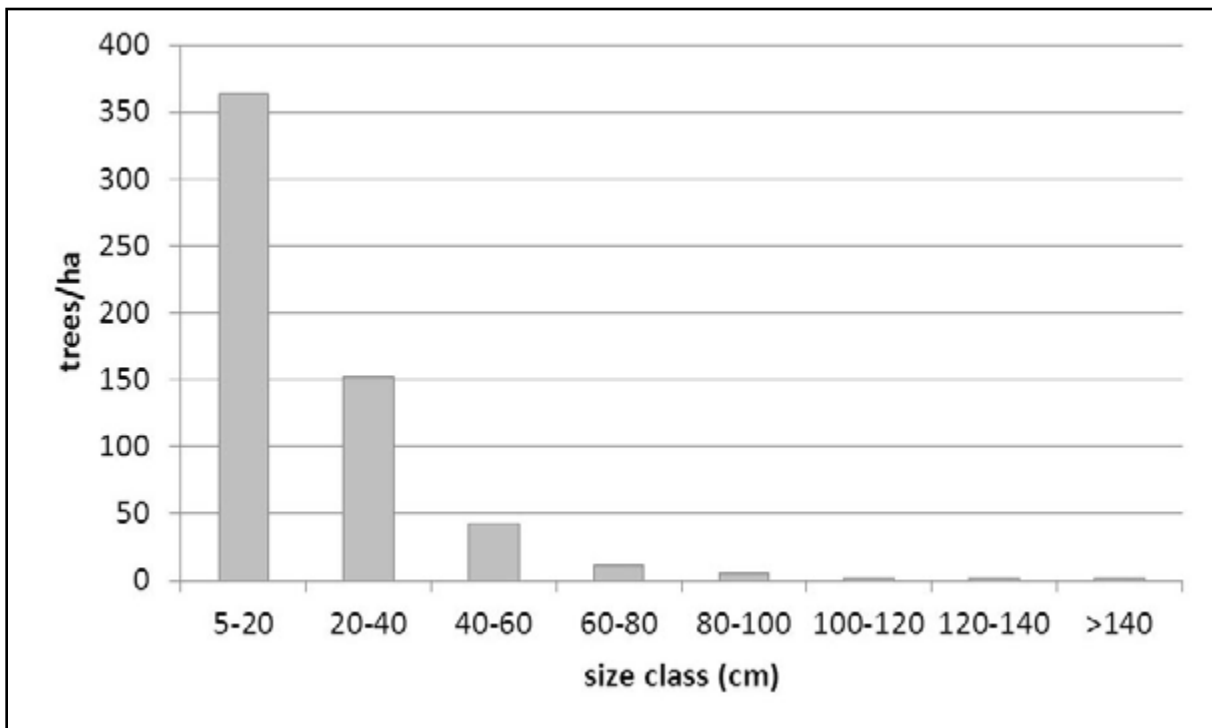


Figure 20. Basal area for reference and current YPMC forests, means of four stand reconstruction studies from the west slope of the Sierra Nevada. Data from Scholl and Taylor 2011, USDA 1911, Parsons and Debenedetti 1979 (yellow pine and mixed conifer sites). See Figure 22 for species codes.

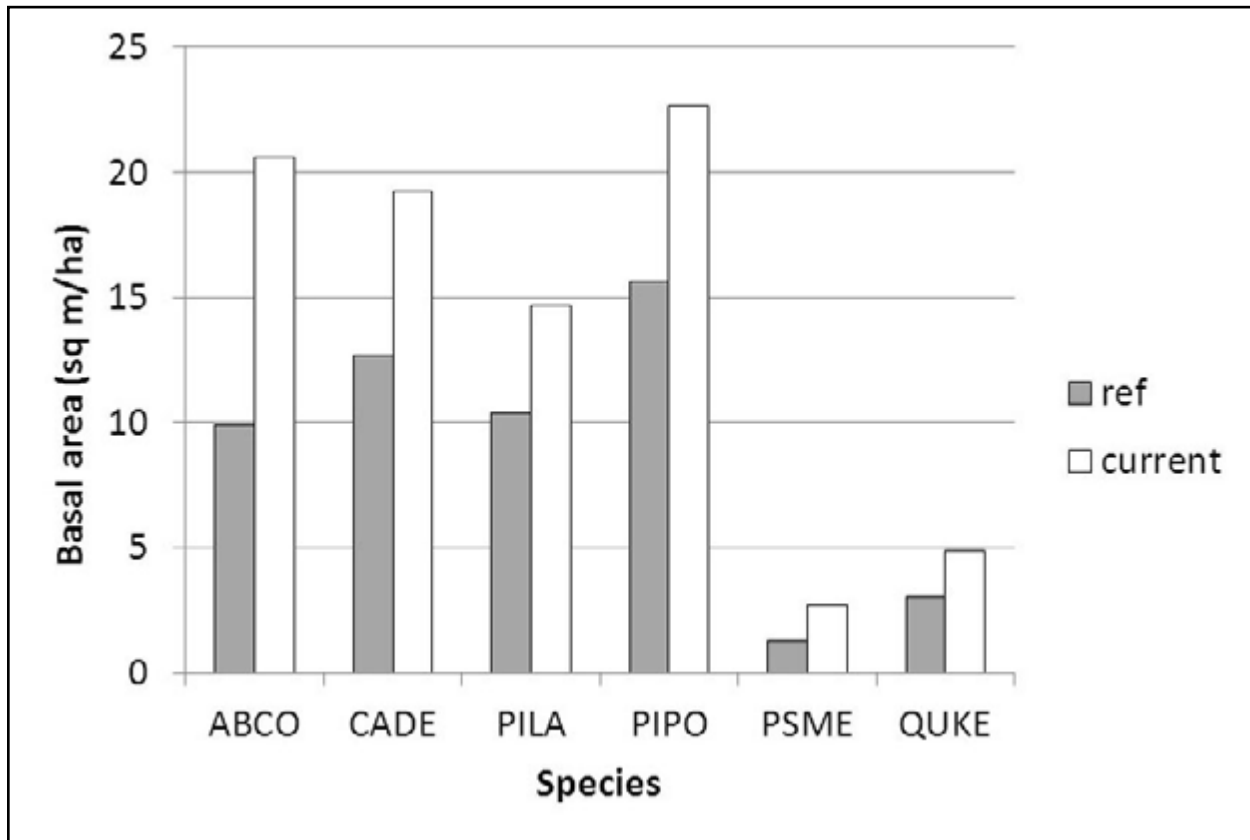


Figure 21. Forest fuels, before and after prescribed fire. Means from 30 late spring and early fall burns studied by Kauffman and Martin (1989) in the northern Sierra Nevada. Error bars are standard errors, based on N = 6 (each sample representing 5 fires)

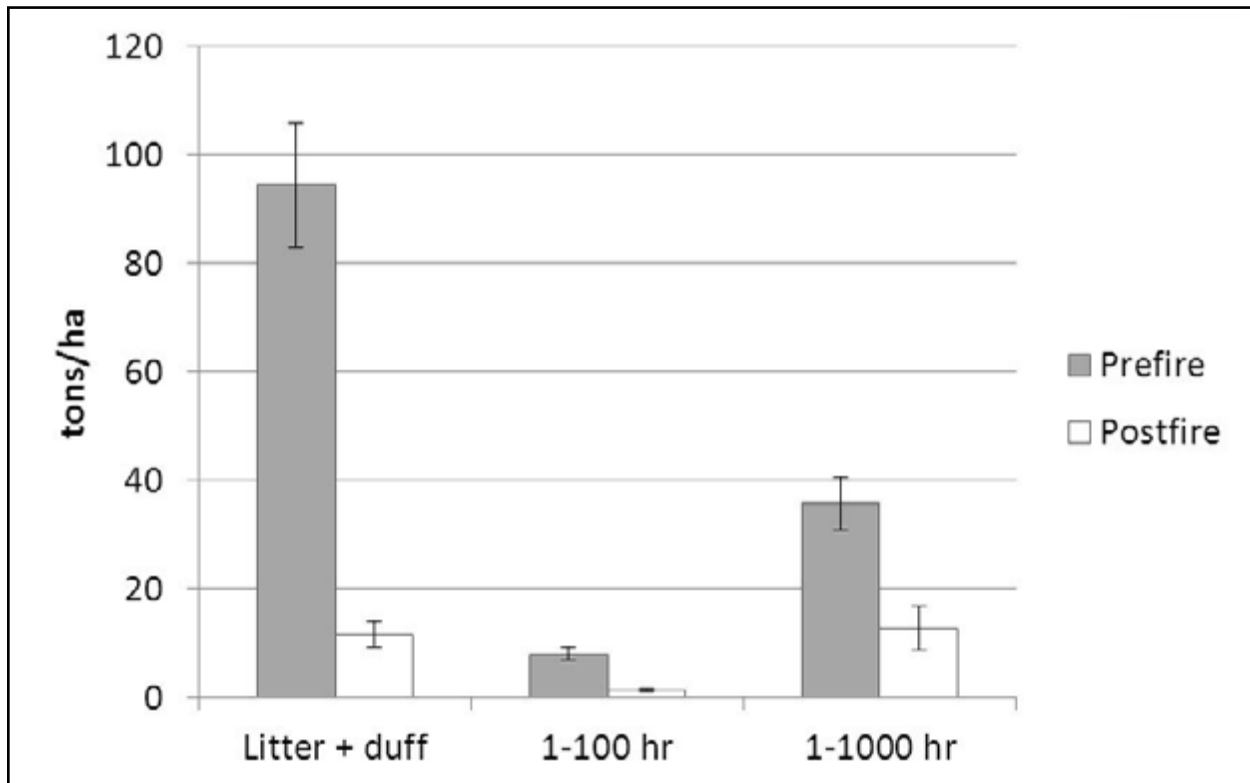


Figure 22. Relative densities of tree species in YPMC forests, comparing VTM (early 1930s) and FIA (early 2000s) data sets in the central assessment area (Plumas, Tahoe, Eldorado, Humboldt-Toiyabe, Eldorado, Stanislaus, and Sierra National Forests, Lake Tahoe Basin Management Unit, and Yosemite National Park; VTM data were not collected north or south of this area). Only trees >10 cm (4") dbh included. ABCO = *Abies concolor* (white fir); ABMA = *A. magnifica* (red fir); CADE = *Calocedrus decurrens* (incense cedar); PIJE = *Pinus jeffreyi* (Jeffrey pine); PILA = *P. lambertiana* (sugar pine); PIPO = *P. ponderosa* (Ponderosa pine); PSME = *Pseudotsuga menziesii* (Douglas-fir); QUKE = *Quercus kelloggii* (black oak); Live oaks = canyon live oak and interior live oak. Data from Dolanc et al. (in prep.).

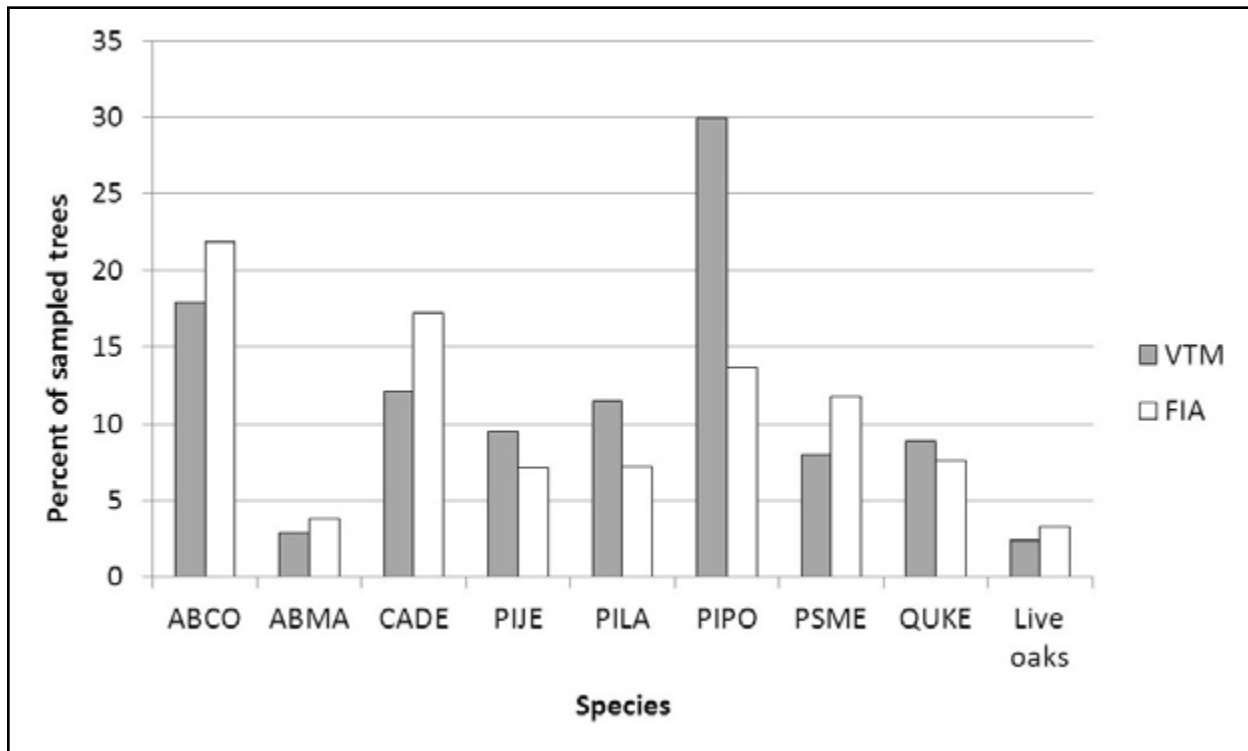


Figure 23. Reconstructions of historical stem densities at seven sites in YPMC forest in the assessment area. Sites arranged in order of proportional dominance by shade intolerant species, from high (left) to low (right). Upper figure portrays absolute densities, lower figure relative densities (percent of stems). 1: Parsons and Debenedetti 1979 (yellow pine, ≥ 12 cm dbh); 2: Taylor 2004 (≥ 10 cm); 3: USDA 1911 (≥ 15.2 cm); 4: North et al. 2007 (≥ 5 cm dbh); 5: Scholl and Taylor 2010 (≥ 10 cm); 6: Taylor, in press (≥ 5 cm dbh); 7: Parsons and Debenedetti 1979 (mixed conifer). See Figure 22 for species codes.

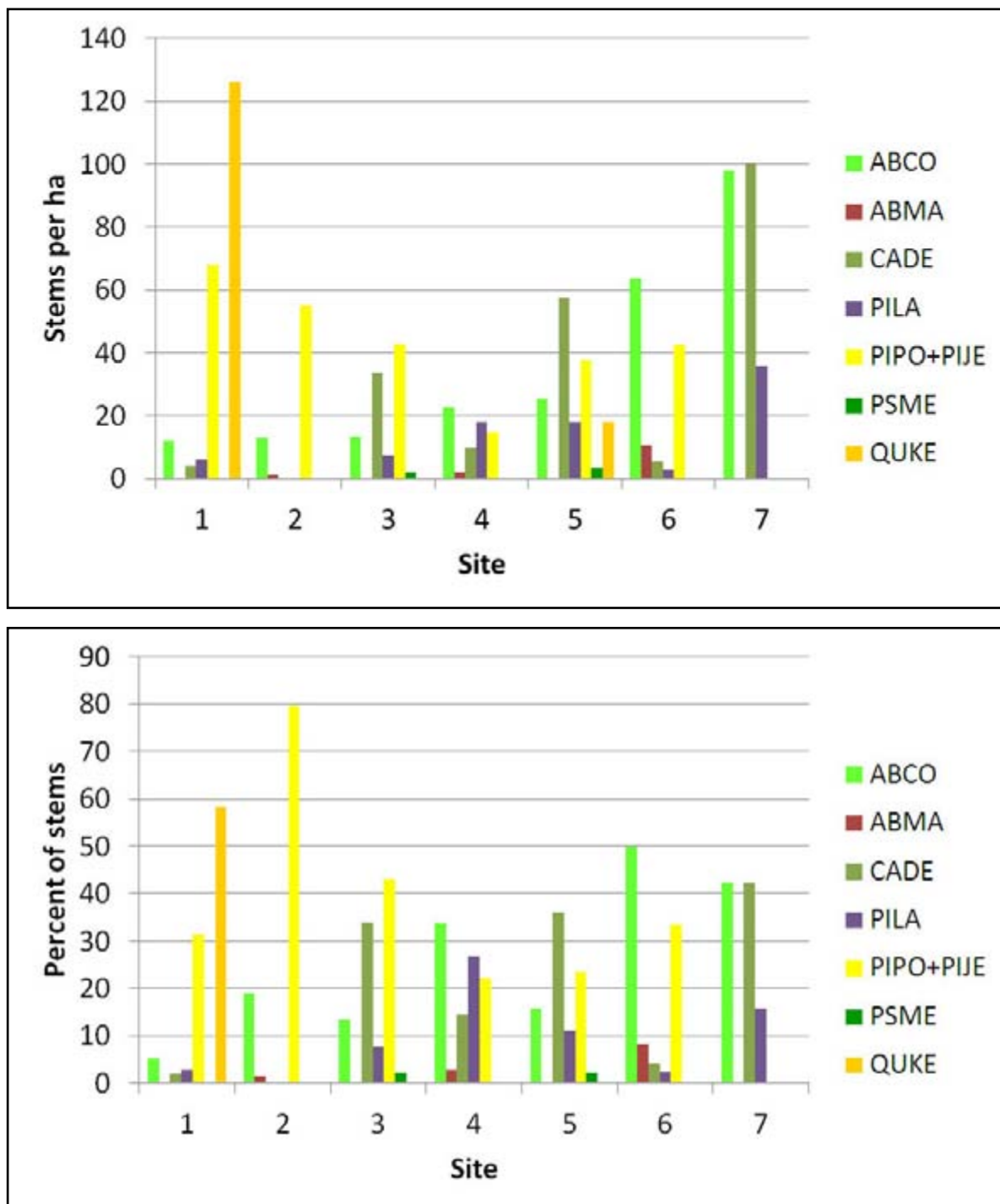


Figure 24. Relative change in species frequencies (measured as the percent of plots in which a given species is found) between the 1930s Forest Service vegetation mapping inventory (“VTM”) and the most recent compilation of FIA data (USFS 2013). Species below the X-axis have dropped in frequency. Codes from the first two letters of the genus plus the first two letters of the species; see Figure 22 for partial list of species codes. Figure 11 from Dolanc et al. (in review)

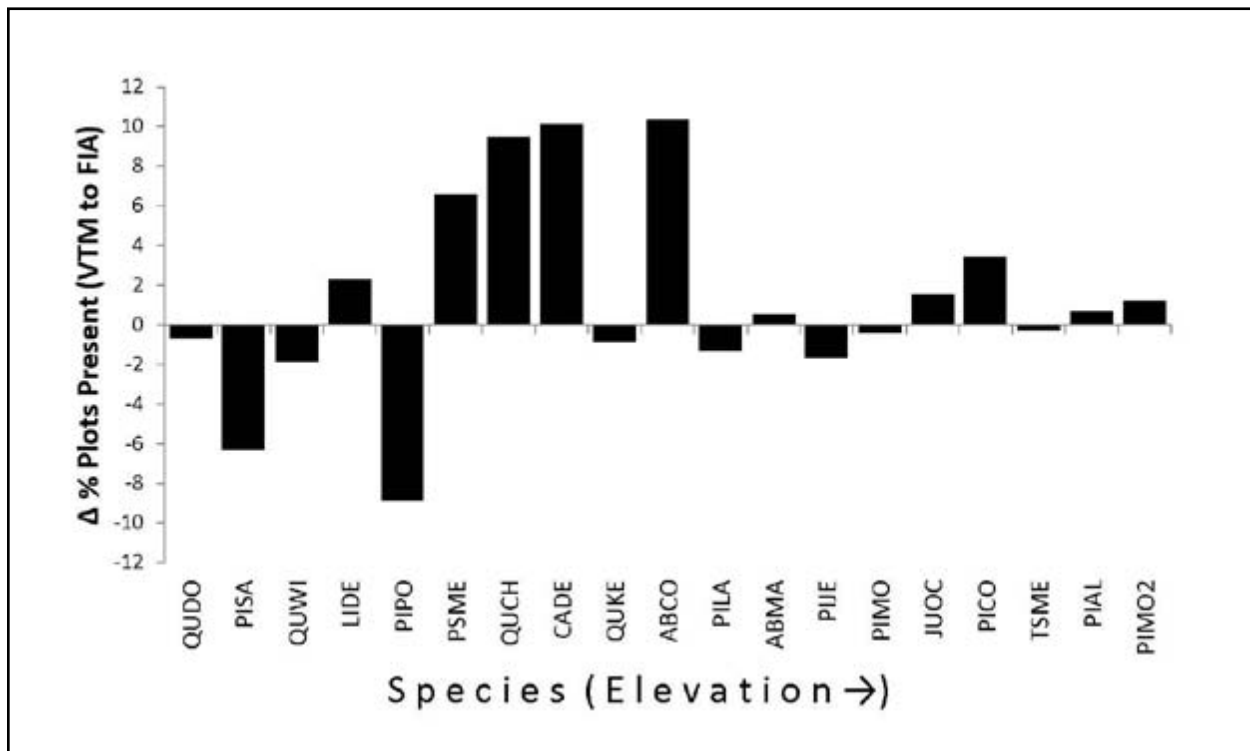
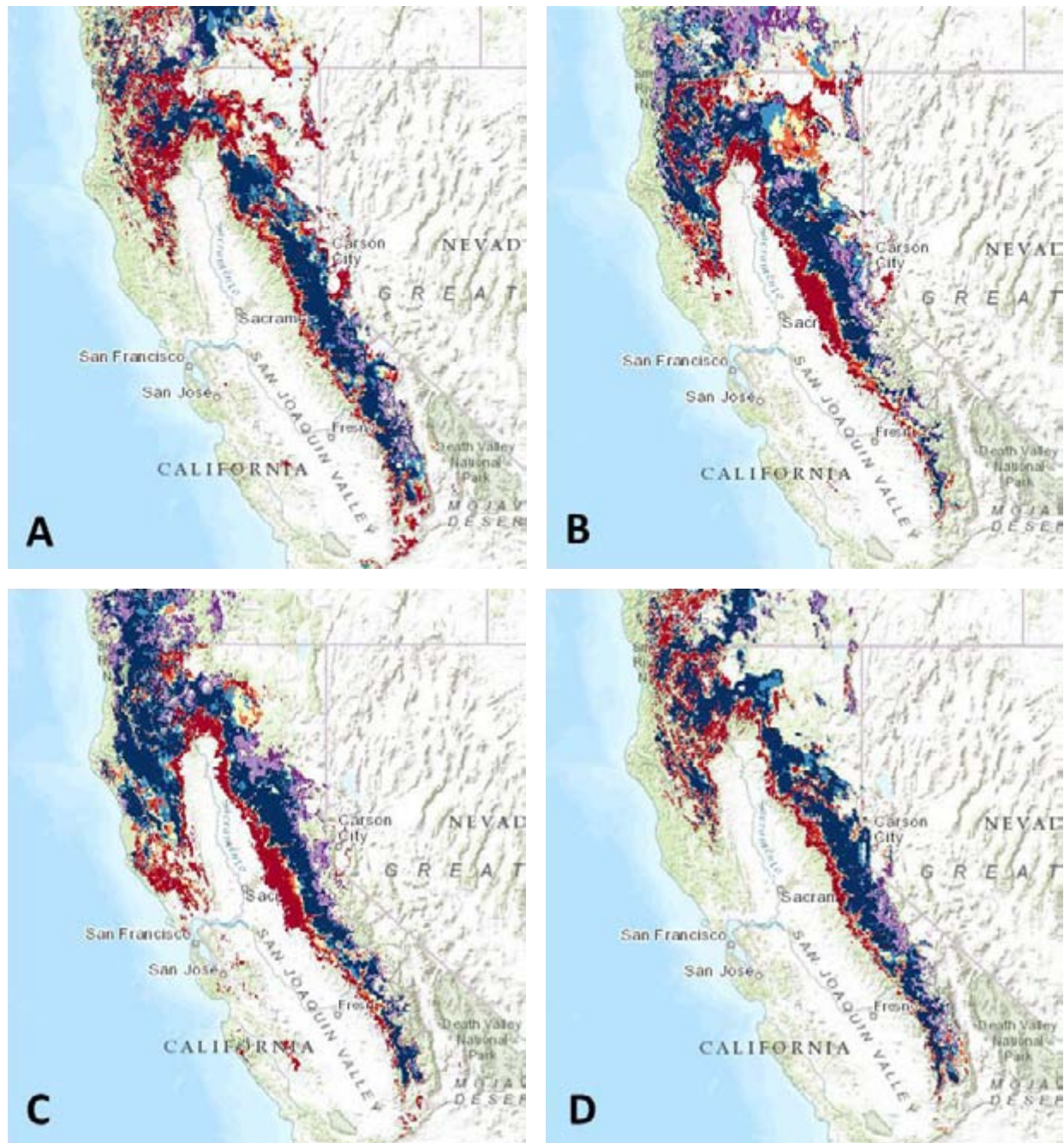


Figure 25. Future climate suitability forecasts for four YPMC tree species for the period 2045-2065, ensemble results from 11 GCMs under the A2 IPCC climate scenario. A – Jeffrey pine; B – ponderosa pine; C – black oak; D – white fir. Red and orange = areas where climates are currently suitable but will not be in the future (climate “stress”); Blues = areas where current and future climates are suitable (refugia); Purples = areas where climates are not currently suitable but will be in the future (possible expansion). Mapped outputs of MaxEnt species distribution projections from modeling by The Nature Conservancy. See text for details.



APPENDIX

Pacific Southwest Region Natural Range of Variation assessments for Forest Planning: A summary of the process and the products

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According to the Forest Service 2012 Planning Rule (36 CFR 219):

“Assessments rapidly evaluate existing information about relevant ecological, economic, and social conditions, trends, and sustainability and their relationship to the land management plan within the context of the broader landscape. The responsible official shall consider and evaluate existing and possible future conditions and trends of the plan area, and assess the sustainability of social, economic, and ecological systems within the plan area, in the context of the broader landscape.”

Planning Rule identifies 15 topic areas that must be addressed in the assessment. Terrestrial ecosystems and system drivers

FSH 1909.12 Chapter 10, Section 12.11

Requires that “the responsible official should identify and evaluate the ecological integrity of... ecosystems within the plan area.”

Steps include:

1. Identify relevant ecosystems to be evaluated and the appropriate scale for the assessment
2. Rapidly evaluate available information about those ecosystems, including composition, structure, and function, by:
 - A. Selecting key ecosystem characteristics for the evaluated ecosystems that will permit evaluation of ecological integrity (sustainability)
 - B. Describe the Natural Range of Variation (NRV) for key ecosystem characteristics, when such information is readily available
 - C. (describes an alternative to NRV assessment)
 - D. Describe current condition and trends of the key ecosystem characteristics
3. Identify and evaluate system drivers and stressors
4. Describe the projected future status of ecosystem integrity, using the key ecosystem characteristics, by:
 - A. Describing the status of the key ecosystem characteristics by comparing NRV to current conditions, or
 - B. (describes alternative where NRV assessment is not available)

5. Identify status of key ecosystem characteristics and determine whether they are “functioning in a way that contributes to ecosystem integrity and sustainability.”

Natural Range of Variation (NRV) assessments

Carried out by Pacific Southwest Region Ecology Program. Each ecologist assigned one or two chapters. Work began October, 2012, completed in May, 2013 (with some exceptions).

Detailed steps:

1. Identify ecosystems to be addressed
 - A. NRV assessments are focused on terrestrial ecosystems
 - B. We used Barbour and Billings (2000) Sugihara et al. (2006), Barbour et al. (2007), MCV, Van de Water and Safford (2011), and California WHR classification, and then requested input from about 25 academic, NGO, and agency experts. Ecosystems had to be mappable, relatively well-studied, common in some form to all of the sources above, and represented through much of the bioregional assessment area. Settled on:
 - i. Forest types
 - a. Yellow pine
 - b. Mixed conifer
 - c. Red fir
 - d. Subalpine
 - e. Pinyon-juniper
 - f. Hardwoods
 - Oak-dominated forests
 - Aspen
 - ii. Shrub types
 - a. Chaparral
 - b. Sagebrush
 - iii. Herbaceous types
 - a. Montane meadows
 - iv. Combined types
 - a. Riparian vegetation
2. Determine appropriate scale
 - A. Spatial:
 - i. SNEP/SNFPA boundary
 - ii. NRV reference data used from outside of area when necessary and reasonable
 - iii. Assessments included both stand-level and landscape-level analyses where possible

- B. Temporal:
 - i. Holocene (12,000 ybp to today), with period 1500-1850 set as principal historical reference period (this is a general HRV/NRV standard)
 - ii. FSH 1902.12 guidance: should be before widespread Euroamerican influence
- 3. Determine key ecosystem characteristics (= ecological indicators)
 - A. Guidance from FSH 1902.12 Chapter 10
 - i. “Key ecosystem characteristics include the dominant ecological characteristics that describe the composition, structure, function, and connectivity of terrestrial, aquatic, and riparian ecosystems that are relevant to addressing important concerns about the land management plan. Key ecosystem characteristics are identified, selected, and evaluated during the assessment phase, brought forward to inform the development of plan components, and may be useful for monitoring progress towards maintaining or restoring ecological integrity. Key ecosystem characteristics may be added or modified during the planning phase.”
 - ii. Need to be characteristics that can be measured, are meaningful, and for which we have data both now and in the past, as well as in contemporary reference ecosystems. Also should be something which will respond to Forest Service management, or “indicates something about the limits to Forest Service authority or the inherent capability of the land.”
 - iii. Some examples provided in FSH 1902.12 Chapter 10 (12.14 exhibit 01)
 - iv. Key ecosystem characteristics include ecological processes (“drivers and stressors”)
 - B. Generated list of ecological indicators (= key ecosystem characteristics)
 - i. Ecological indicators: “measurable characteristics of the structure (e.g., genetic, population, habitat, and landscape pattern), composition (e.g., genes, species, populations, communities, and landscape types), or function (e.g., genetic, demographic/life history, ecosystem, and landscape disturbance processes) of ecological systems.” (Niemi and McDonald. 2004. ARES 35: 89-111)
 - ii. Based on nested hierarchy of:
 - a. Ecosystem attribute (highest level)
 - Composition
 - Structure
 - Function
 - b. Ecological hierarchy
 - Population/species
 - Community/ecosystem

- Landscape/region
- c. Indicator group (examples)
 - Species diversity
 - Physiognomy
 - Productivity
 - Biogeography
 - Nutrient cycling
 - Patchiness
 - Connectivity
 - Disturbance
 - ◊ Fire, grazing, logging, wind, floods, etc.
- d. Indicators
 - Examples from fire regime indicator group
 - ◊ Fire frequency, fire rotation, fire size, fire severity, fire season...
 - Examples from physiognomy indicator group
 - ◊ Tree density, basal area/volume, canopy cover...
- e. Variables and their units (lowest level)
 - Example from fire frequency indicator
 - ◊ Fire return interval (mean number of years between fires)
 - Example from tree density indicator
 - ◊ Number of trees per hectare
- f. A spreadsheet of attributes, hierarchy, indicators and variables is available from the Regional Ecologist (but perusal of the chapters will also provide this list)

4. NRV assessments

- A. 11 Chapters, Introductory chapter to be written; intention is to publish as General Technical Report in late 2013
- B. NRV is based on historical and contemporary reference systems. Direct data used when available, inference used where necessary and justifiable
- C. Includes comparisons to current conditions and summary of literature re. possible future trends
- D. Focus is on peer reviewed publications, including papers in press or soon to be in press; government publications; Forest Service and other federal and state agency data; and in some cases academic theses or dissertations. Because information on the historical state of some ecosystems and ecological processes and patterns is

scarce, in some cases we also refer to published anecdotal information from the mid-19th to early 20th centuries. We do not refer to anecdotal information from more recent times.

- E. Used Forest Service Rocky Mountain Region HRV assessments from early 2000s as rough templates (e.g., Dillon et al. 2005, Meyer et al. 2005, Veblen and Donnegan 2005)
- F. Basic outline
 - i. Introduction
 - ii. Methods
 - iii. NRV descriptions
 - a. Function
 - b. Structure
 - c. Composition
 - iv. Summary of NRV deviations
 - v. Literature cited
 - vi. Tables and figures
- G. Timeline
 - i. Dec 14: Preliminary bibliography assembled
 - ii. Feb 11: Drafts due for internal review
 - iii. Feb 15: Internal reviews due
 - iv. Feb 25: Delivery of draft chapters to Planning staff
 - v. Mar 11: Final revised drafts due for external review
 - vi. Apr 1: External reviews due
 - vii. Week of Apr 8: Final draft NRV assessments
 - viii. Week of May 12: Final NRV assessments posted to web